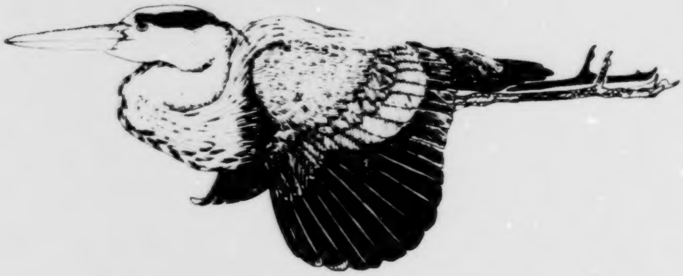
Environment CANADA Environnement
2046080G VOL ISS 19 1987
EXC # 001
SK 470 T42
TECHNICAL REPORT SERIES (CANADIAN WILDLIFE SERVICE)
OOFF

THE IMPACT OF PESTICIDES ON THE ECOLOGY OF PRAIRIE NESTING DUCKS

P.J. Sheehan
A. Baril
P. Mineau
D.K. Smith
A. Harfenist
W.K. Marshall



TECHNICAL REPORT SERIES No. 19
Headquarters 1987
Canadian Wildlife Service

Environment Canada
Environnement Canada

Canadian Wildlife Service
Service canadien de la faune

Canada

87-06089/

TECHNICAL REPORT SERIES
CANADIAN WILDLIFE SERVICE

This series of reports, established in 1986, contains technical and scientific information from projects of the Canadian Wildlife Service. The reports are intended to make available material that either is of interest to a limited audience or is too extensive to be accommodated in scientific journals or in existing CWS series.

Demand for these Technical Reports is usually confined to specialists in the fields concerned. Consequently, they are produced regionally and in small quantities; they can be obtained only from the address given on the back of the title page. However, they are numbered nationally. The recommended citation appears on the title page.

Technical Reports are available in CWS libraries and are listed with the DOBIS system in major scientific libraries across Canada. They are printed in the official language chosen by the author to meet the language preference of the likely audience. **To determine whether there is significant demand for making the reports available in the second official language, CWS invites users to specify their official language preference. Requests for Technical Reports in the second official language should be sent to the address on the back of the title page.**

SÉRIE DE RAPPORTS TECHNIQUES
DU SERVICE CANADIEN DE LA FAUNE

Cette série de rapports donnant des informations scientifiques et techniques sur les projets du Service canadien de la faune (SCF) a démarré en 1986. L'objet de ces rapports est de promouvoir la diffusion d'études s'adressant à un public restreint ou trop volumineuses pour paraître dans une revue scientifique ou l'une des séries du SCF.

Ordinairement, seuls les spécialistes des sujets traités demandent ces rapports techniques. Ces documents ne sont donc produits qu'à l'échelon régional et en quantités limitées; ils ne peuvent être obtenus qu'à l'adresse figurant au dos de la page titre. Cependant, leur numérotage est effectué à l'échelle nationale. La citation recommandée apparaît à la page titre.

Ces rapports se trouvent dans les bibliothèques du SCF et figurent aussi dans les listes du système de référence DOBIS utilisé dans les principales bibliothèques scientifiques du Canada. Ils sont publiés dans la langue officielle choisie par l'auteur en fonction du public visé. **En vue de déterminer si la demande est suffisamment importante pour produire ces rapports dans la deuxième langue officielle, le SCF invite les usagers à lui indiquer leur langue officielle préférée. Il faut envoyer les demandes de rapports techniques dans la deuxième langue officielle à l'adresse indiquée au verso de la page titre.**

Cover illustration is by R.W. Butler and may not be used for any other purpose without the artist's written permission.

L'illustration de la couverture est une œuvre de R.W. Butler. Elle ne peut dans aucun cas être utilisée sans avoir obtenu préalablement la permission écrite de l'auteur.

THE IMPACT OF PESTICIDES ON THE ECOLOGY OF PRAIRIE NESTING DUCKS

P.J. Sheehan
A. Baril
P. Mineau
D.K. Smith
A. Harfenist
W.K. Marshall

Technical Report Series No. 19
Headquarters 1987
Canadian Wildlife Service

This publication may be cited as:

Sheehan, P.J., Baril, A., Mineau, P., Smith, D.K., Harfenist, A., and Marshall, W.K. The impact of pesticides on the ecology of prairie nesting ducks. Technical Report Series No. 19, Canadian Wildlife Service, Headquarters

Issued under the authority of the
Minister of the Environment
Canadian Wildlife Service

©Minister of Supply and Services Canada 1987
Catalogue No. CW69-5/19E
ISBN 0-662-15358-8
ISSN 0831-6481

Copies may be obtained from:

Wildlife Toxicology and Surveys Branch
National Wildlife Research Centre
Canadian Wildlife Service
Environment Canada
Ottawa
K1A 0H3

FOREWORD

Because of its size and complexity, this report has now been some time in the making. This exercise was begun in the fall of 1983, prompted by requests for the broadscale aerial use in cereal crops of a synthetic pyrethroid, deltamethrin. What was initially intended as a simple review of the possible ecological impacts of such a registration quickly grew to become a complete assessment of the prairie pothole ecosystem under siege by modern agricultural development with, emphasis on pesticide use.

The bibliographic research for this exercise took place in 1984 and 1985. The last pesticide use information collected was for the 1985 spray season although even some 1985 information was not included in the main body of the report because it was received too late (e.g. Appendix 4.1). Likewise, the information on bird kills (Appendix 1.1) has now been substantially updated as a result of more recent information (Canadian Wildlife Service, unpublished).

As we go to press, three separate publications have appeared which are based wholly or in part on information generated in this report:

Shaw, G.G., D.K. Smith, P.J. Sheehan and P. Mineau. 1984. Environmental concerns. (In) P.W. Voisey [ed.] Symposium on the future role of aviation in agriculture. National Research Council of Canada. AFA-TN-M (NRC 23504), pp. 47-63.

Grue, C.E., L.R. DeWeese, P. Mineau, G.A. Swanson, J.R. Foster, P.M. Arnold, J.N. Huckins, P.J. Sheehan, W.K. Marshall and A.P. Ludden. 1986. Potential impacts of agricultural chemicals on waterfowl and other wildlife inhabiting prairie wetlands: an evaluation of research needs and approaches. Transactions of the 51st North American and Natural Resources Conference. Wildlife Mgmt. Inst. Washington D.C. pp. 357-383.

Mineau, P., P.J. Sheehan and A. Baril. In Press. Pesticides and waterfowl on the canadian prairies: A pressing need for research and monitoring. (In) A.W. Diamond and F. Filion [eds.] The values of birds. I.C.B.P. Technical publication No.___. Cambridge, U.K.

Perhaps more importantly, this report will serve as a framework under which the evaluation of pesticides in use or proposed for use in the canadian prairies and, to some extent in the rest of Canada, will be evaluated.

It is impossible, without missing anyone, to acknowledge all the individuals who responded to our queries over the phone or in person or those that were kind enough to review parts of this document. Therefore, we will not attempt to do so. A number of individuals provided unpublished information and their contribution is aknowledged throughout the body of the report. Any errors in the transcription or interpretation of the published or unpublished data are strictly the responsibility of the authors. To everyone involved: our most sincere thanks. We would be remiss however if we did not acknowledge Suzanne Vincent and other members of the typing pool who patiently typed and re-typed this report over the 3+ years of production.

For more information as well as any clarifications on the report, enquiries should be addressed to P. Mineau, Wildlife Toxicology and Surveys Branch, Canadian Wildlife Service, Ottawa, K1A 0H3.

Abstract

Approximately one-quarter of the total North American duck population breeds on the Canadian Prairies. Encroachment of prairie agriculture on waterfowl nesting habitat has been extensive and shows no sign of abating. Whereas this physical disruption of wildlife habitat is all too visible, the effects of pesticides on the quality of waterfowl habitat and directly on waterfowl are not so readily apparent and have largely been ignored.

This report is intended to fill that gap. In it we review in detail the ecology of prairie sloughs and waterfowl, assess the overlap between waterfowl nesting and pesticide use, and examine the main threats associated with both insecticides and herbicides as well as the more generic problem of off-site contamination. Finally, we attempt to put all that information together in a series of scenarios and predict the likely impacts of actual pesticide use. Chapter 9 of the report serves as an executive summary.

We conclude that insufficient attention has been paid to the impacts of pesticide use. Our various scenarios, some of which are considered extremely conservative, indicate that the number of breeding waterfowl at risk and the possible loss of waterfowl broods are in the same range as the impact on waterfowl from the recreational hunting harvest in the three Prairie Provinces. Given current concerns over declining stocks, it is important to stop ignoring this aspect of prairie agriculture.

Résumé

Environ un quart de la population nord-américaine de canards utilisent les prairies canadiennes comme lieu de reproduction. L'empiétement de l'agriculture sur l'aire de nidification de la sauvagine dans les Prairies est considérable et ne montre pas de signes d'essoufflement. Alors que cette destruction physique des habitats propices à la faune est partout évidente, l'impact plus insidieux des pesticides sur la qualité de l'habitat de la sauvagine et plus directement sur celle-ci a été dans une large mesure négligé jusqu'à ce jour.

Le présent rapport entend combler ce vide. Le lecteur y trouvera une revue détaillée de l'écologie des terres marécageuses et des populations de canards des prairies, la mesure de l'empiétement des pesticides sur les aires de nidification de la sauvagine, une évaluation des principales menaces causées par l'utilisation des insecticides et herbicides et un examen du problème de la contamination hors des sites d'épandage. Suit enfin une synthèse de toute cette information sous forme de scénarios visant à définir l'impact possible de l'utilisation réelle des pesticides. Le chapitre 9 constitue un résumé détaillé du rapport.

La conclusion souligne la négligence démontrée dans le fait de ne pas avoir davantage tenu compte de l'impact de l'utilisation des pesticides. Les scénarios présentés ici (dont certains sont considérés comme étant très prudents) démontrent que le nombre de canards reproducteurs exposés à un risque ainsi que la perte de production possible associée à l'utilisation de pesticides sont de même envergure que l'impact des prises faites dans le cadre de la chasse sportive pour les trois provinces des Prairies. Compte tenu des préoccupations actuelles causées par la diminution des effectifs de canards, il n'est plus possible d'ignorer plus longtemps cet aspect de l'exploitation agricole des Prairies.

Pour toute question relative à ce rapport, prière de s'adresser à: Pierre Mineau, Direction des inventaires et de la toxicologie de la faune, Service Canadien de la faune, Ottawa K1A 0H3

TABLE OF CONTENTS

PAGE

FOREWORD I

ABSTRACT III

RESUME IV

TABLE OF CONTENTS V

LIST OF FIGURES XII

LIST OF TABLES XIV

PERSONNAL COMMUNICATIONS XXIII

Chapter 1 INTRODUCTION

Appendix 1.1 The toxicity of current prairie pesticides to waterfowl. A1.1
  Table A: Acute oral toxicity of selected insecticides to Mallard Ducks
  Table B: Toxicity values for Mallard ducklings exposed to selected insecticides via dietary intake
  Table C: Effects of insecticides and herbicides in aqueous emulsion on Mallard embryos
  Table D: Acute oral toxicity of selected herbicides to Mallard Ducks
  Table E: Toxicity values for Mallard ducklings were exposed to selected herbicides via dietary intake

Chapter 2 WATERFOWL AND PRAIRIE SLOUGH ECOLOGY

2.1 Physical Aspects of the Prairie Pothole Region 2.1
2.1.1 Geology and soils of the prairie pothole region 2.1
2.1.2 Climate 2.2
2.1.3 Major vegetation zones 2.2
2.1.4 Pothole distribution 2.2
2.1.5 Water regime 2.4

2.2 Water and Soil Chemistry of Potholes 2.6
2.2.1 Water chemistry 2.6
2.2.2 Soil chemistry 2.7

2.3 The Vegetation In and Around Potholes 2.8
2.3.1 Vegetation zones 2.9
2.3.1.1 Emergent species 2.9
2.3.1.2 Submergent and floating species 2.10
2.3.1.3 Algae 2.10
2.3.2 Factors affecting plant distribution within potholes 2.11
2.3.2.1 Water regime 2.12
2.3.2.2 Salinity 2.14

TABLE OF CONTENTS (CONT'D)

PAGE

2.3.2.3 Disturbance 2.14
2.3.2.4 Soil properties 2.15
2.3.2.5 Consumers 2.15
2.3.3 Nutrients and production 2.16
2.3.3.1 Nutrient import and export 2.16
2.3.3.2 Compartmentalization of nutrients 2.17
2.3.3.3 Cycling processes 2.17
2.3.3.4 Production 2.19

2.4 The Invertebrates of Prairie Sloughs 2.22
2.4.1 Invertebrate abundance and distribution 2.23
2.4.1.1 Plant-invertebrate associations 2.24
2.4.1.2 Variation between potholes 2.27
2.4.1.3 Variability between years 2.27
2.4.1.4 Seasonal patterns of abundance 2.28
2.4.2 Factors affecting invertebrate assemblages 2.29
2.4.2.1 Water regime 2.29
2.4.2.2 Salinity 2.30
2.4.2.3 Agriculture 2.30
2.4.3 Trophic relationships 2.30
2.4.3.1 Physical and chemical effects 2.30
2.4.3.2 Foods of invertebrates 2.31
2.4.3.3 The detrital pathway 2.32

2.5 Migratory Ducks and Prairie Wetland Habitat 2.35
2.5.1 Chronology of life history events during reproduction 2.35
2.5.2 Habitat requirements 2.37
2.5.2.1 General needs 2.37
2.5.2.2 Nesting habitat 2.39
2.5.2.3 Brood vulnerability to predators 2.44
2.5.3 Food resources during reproduction 2.45
2.5.3.1 Laying female 2.45
2.5.3.2 Ducklings 2.49
2.5.3.3 Quantity of food resources 2.51
2.5.4 Indices of reproductive success 2.52
2.5.4.1 Pair and nest density 2.53
2.5.4.2 Clutch size and percent hatching succes 2.55
2.5.4.3 Brood success and recruitment 2.56
2.5.4.4 Models and recruitment estimates 2.57

Chapter 3 WATERFOWL HABITAT DISTRIBUTION AND AGRICULTURE OVERLAP

3.1 Waterfowl Numbers and Distribution 3.1

3.2 Quantitative Analysis of Waterfowl Habitat in the Prairies 3.4
3.2.1 The Canada Land Inventory (CLI) 3.5
3.2.2 The CLI waterfowl data 3.6
3.2.3 Validation of the CLI waterfowl data 3.7
3.2.4 Current status of prairie waterfowl habitat 3.8

TABLE OF CONTENTS (CONT'D)

PAGE

3.3 Agricultural Land Use in the Prairies 3.11
3.3.1 CLI agriculture potential and actual agricultural use 3.11
3.3.2 Crop type and distribution 3.12

3.4 Agriculture-Waterfowl Overlap 3.13
3.4.1 Analysis of the overall prairie overlap 3.14
3.4.2 Regional analysis of overlap 3.15
3.4.3 Inter-year variation in agriculture-waterfowl overlap and waterfowl production 3.17

Appendix 3.1 Adjusted breeding populations means (for the 1970-1983 period) for all duck species surveyed annually by stratum.

Appendix 3.2 Canadian and U.S. prairie (grassland and parkland) subtotals of mean estimated breeding duck populations (1970-1983).

Appendix 3.3 Provincial and state subtotals of mean estimated breeding duck populations.

Appendix 3.4 Mean estimated breeding duck population (1970-1983) broken down by major vegetation zone.

Appendix 3.5 Comparison of the full Canada Land Inventory (CLI) data base with the more restricted data base corresponding to the prairies (grasslands and parklands) following Patterson (1978).

Appendix 3.6 Land areas classified by agricultural class within the parkland-grassland area as defined by Patterson (1978) compared to areas classified under the full CLI data base.

Appendix 3.7 "Present land use" compilation of data classified under the CLI (1965 data).

Appendix 3.8 Agriculture-waterfowl land capability cross tabulation by primary CLI agriculture and waterfowl classification.

Appendix 3.9 Estimated number of diving and dabbling ducks nesting within a number of major crop types (or within margins associated with those crops) by census division.

Chapter 4 PESTICIDE USE IN THE PRAIRIES

4.1 The Compilation of Pesticide Use Statistics 4.1
4.1.1 Pesticide surveys 4.1
4.1.1.1 Federal statistics 4.1
4.1.1.2 Alberta 4.3
4.1.1.3 Saskatchewan 4.3

TABLE OF CONTENTS (CONT'D)

PAGE

4.1.1.4 Manitoba 4.4
4.1.2 Surveys of pest control programs 4.4

4.2 Insecticide Use and Overall Market Share 4.4
4.2.1 Grasshopper control 4.6
4.2.2 Other insect control programs and total insecticide use 4.8
4.2.3 Temporal overlap between insecticide use and duck breeding 4.10
4.2.4 Factors affecting insecticide use 4.10
4.2.4.1 Proximate factors 4.10
4.2.4.2 Climate, geography and grasshoppers 4.11
4.2.4.3 Ultimate factors, cost benefit of insecticide use 4.11

4.3 Herbicide Use 4.12
4.3.1 Types and quantities of herbicide use 4.12
4.3.2 Market share data 4.13
4.3.3 Temporal overlap between herbicide use and duck breeding 4.14
4.3.4 Factors affecting herbicide use 4.15
4.3.4.1 Pest distribution 4.15
4.3.5 Ultimate factors, cost-benefit of herbicide use 4.15

4.4 Other Chemical Uses 4.16

Appendix 4.1 Recent (1985) information from the Manitoba Crop Insurance Corporation on market share data for herbicides used in Manitoba.

Appendix 4.2 Insecticides recommended in 1984 for selected crops and pests in three prairie provinces.

Appendix 4.3 Herbicides recommended by one or more provinces between 1982 and 1985, primarily for the control of a) broadleaf weeds, b) grassy weeds, c) both broadleaf and grassy weeds or as dessicant

Appendix 4.4 Example of the information available through the annual registrant survey published by Manitoba on an annual basis.

Appendix 4.5 Distribution maps for the major broadleaf and grassy weeds in all three prairie provinces.

Chapter 5 LIKELIHOOD OF PESTICIDE CONTAMINATION OF WETLANDS

5.1 Pathway of Entry 5.1

5.2 Control of Deposit 5.3
5.2.1 Factors influencing pesticide drift 5.4
5.2.2 Factors specific to prairie agriculture 5.6
5.2.2.1 Formulation and final tank mix 5.6
5.2.2.2 Application practices 5.7
5.2.2.3 Meteorological 5.7

TABLE OF CONTENTS (CONT'D)

PAGES

5.2.2.4 Topographical 5.7

5.3 Analysis of Potential Slough Contamination from Direct Sources 5.7
5.3.1 Overspray 5.8
5.3.2 Drift droplet deposition 5.10

5.4 Contamination from Indirect Sources 5.11
5.4.1 Vapour drift 5.12
5.4.2 Surface runoff 5.12

5.5 Likelihood Summary 5.15

Chapter 6 TOXICITY OF INSECTICIDES IN SLOUGH ECOSYSTEMS: INDIRECT THREAT TO WATERFOWL

6.1 The Comparative Toxicity of Pyrethroid and Other Major Use Insecticides to Aquatic Organisms 6.1
6.1.1 Aquatic insects 6.1
6.1.2 Other aquatic invertebrates 6.3
6.1.3 Chronic toxicity 6.5
6.1.4 Summary based on laboratory toxicity data 6.5

6.2 Model Ecosystems and Field Studies of Insecticide Impacts 6.6
6.2.1 Impacts of pyrethroid insecticides on aquatic ecosystems 6.7
6.2.1.1 Cypermethrin 6.7
6.2.1.2 Deltamethrin 6.8
6.2.1.3 Fenvalerate 6.9
6.2.1.4 Permethrin 6.9
6.2.2 Impacts of carbamate, organophosphorous and organochlorine insecticides on aquatic ecosystems 6.11
6.2.2.1 Carbamates 6.11
6.2.2.2 Organophosphates 6.12
6.2.2.3 Organochlorines 6.15
6.2.3 Critique of field studies 6.15

6.3 Ecological Hazard Evaluation 6.16
6.3.1 Need for a framework to evaluate hazards from pesticides 6.16
6.3.2 Hazard and risk modelling: objectives and evaluation criteria 6.17
6.3.3 Calculation models for the analysis of ecological hazard 6.20
6.3.4 Description of the exposure model component 6.21
6.3.5 Description of the toxicity model component 6.23
6.3.5.1 Integrated exposure model 6.24
6.3.5.2 Maximum concentration model 6.25
6.3.5.3 Choice of test species 6.26
6.3.6 Benchmarks and an index of secondary impacts 6.27
6.3.7 Input data for risk calculations 6.28
6.3.8 Results from relative hazard analysis 6.29
6.3.8.1 Persistence 6.29
6.3.8.2 Predicted hazard to aquatic invertebrates 6.33
6.3.9 Empirical analysis of the influence of physico-chemical data on predicted hazard 6.35

TABLE OF CONTENTS (CONT'D)

PAGES

6.3.10 Benchmark calibration 6.37
6.3.11 Critique of relative hazard analysis 6.38
6.3.12 Hazard to ducklings 6.39
6.3.12.1 Experimental evidence 6.39
6.3.12.2 Duckling growth models 6.41
6.3.13 Critique of secondary impact model applied to ducklings 6.46

6.4 Threat of Synthetic Pyrethroid and Other Insecticides to Waterfowl 6.47

Appendix 6.1 Physico-chemical parameters used for persistence and risk calculations (Tables 1-13).

Chapter 7 TOXICITY OF HERBICIDES IN SLOUGH ECOSYSTEMS AND UPLAND HABITAT: INDIRECT THREAT TO WATERFOWL

7.1 Expected Exposure Pattern in Herbicide Contaminated Sloughs 7.2
7.1.1 Factors affecting herbicide fate in pothole ecosystems 7.3
7.1.2 Partitioning of herbicides in slough ecosystems 7.5
7.1.2.1 Physico-chemical information 7.5
7.1.2.2 Laboratory and field studies 7.7
7.1.2.3 Partitioning summary 7.9
7.1.3 Persistence 7.10
7.1.3.1 Less persistent herbicides 7.10
7.1.3.2 More persistent herbicides 7.11
7.1.4 Exposure analysis 7.14

7.2 Herbicide Effects on Vegetation 7.14
7.2.1 Direct toxicity 7.15
7.2.1.1 Algae 7.15
7.2.1.2 Macrophytes 7.17
7.2.2 Indirect and long-term impact 7.19
7.2.2.1 Ecosystem response 7.19
7.2.2.2 Factors determining community response 7.22
7.2.2.3 Scenarios for prairie potholes 7.23
7.2.3 Problems with data sets 7.24

7.3 Herbicide Damage to Waterfowl Food Resources 7.25
7.3.1 Herbicide impact on invertebrate prey 7.25
7.3.1.1 Laboratory toxicity data 7.25
7.3.1.2 Field studies 7.28
7.3.2 Herbicide impact on plant foods 7.30

7.4 Herbicide Damage to Nesting Habitat 7.33

Chapter 8 HAZARD EVALUATION

8.1 Introduction 8.1

8.2 Herbicides 8.3
8.2.1 Hazard scenarios; input 8.3
8.2.2 Hazard scenarios; output 8.6

TABLE OF CONTENTS (CONT'D)

PAGES

8.2.2.1 No herbicide contamination of margins or "baseline" scenario 8.6
8.2.2.2 Herbicide contamination of margins 8.6
8.2.3 Discussion 8.7

8.3 Insecticides 8.8
8.3.1 Hazard scenarios; input 8.8
8.3.2 Hazard scenarios; output 8.10
8.3.2.1 Annual insecticide use; non-outbreak situation 8.10
8.3.2.2 A major grasshopper outbreak 8.11
8.3.2.3 An outbreak of a major canola pest 8.12
8.3.3 Discussion 8.13

8.4 Conclusion 8.13

Chapter 9 SUMMARY, RECOMMENDATIONS AND RESEARCH NEEDS

9.1 Waterfowl and Prairie Slough Ecology 9.1
9.1.1 Summary 9.1
9.1.2 Recommendations and research needs 9.5

9.2 Waterfowl Habitat Distribution and Agriculture Overlap 9.5
9.2.1 Summary 9.5
9.2.2 Recommendations and research needs 9.8

9.3 Pesticide Use in the Prairies 9.9
9.3.1 Summary 9.9
9.3.2 Recommendations and research needs 9.11

9.4 Likelihood of Pesticide Contamination of Wetlands 9.12
9.4.1 Summary 9.12
9.4.2 Recommendations and research needs 9.13

9.5 Toxicity of Insecticides in Slough Ecosystems: Indirect Threat to Waterfowl 9.14
9.5.1 Summary 9.14
9.5.2 Recommendations and research needs 9.17

9.6 Toxicity of Herbicides in Slough Ecosystems and Upland Habitat: Indirect Threat to Waterfowl 9.18
9.6.1 Summary 9.18
9.6.2 Recommendations and research needs 9.21

9.7 Hazard Evaluation 9.22
9.7.1 Summary 9.22

9.8 Concluding Remarks 9.25

BIBLIOGRAPHY B.1

LIST OF FIGURES

FIGURE	FOLLOWING PAGE
2.1 Prairie pothole region of central North America	2.1
2.2 Soil zones of the prairie pothole region of Canada	2.1
2.3 Major vegetation zones of the prairie provinces of Canada	2.2
2.4 Basin size distribution in three study areas of Saskatchewan	2.3
2.5 Relation of the direction of groundwater flow and water-table configuration	2.6
2.6a Vegetation patterns in stable wetlands shown in vertical and profile diagrams	2.10
2.6b Periods of greatest abundance of dominant invertebrate taxa in four wetland ecosystems	2.28
2.7 Chronology of arrival and nest initiation for the more common duck species in the Canadian prairies	2.35
2.8 Chronology of brood hatch and fledge for the more common duck species in the Canadian prairies	2.37
2.9 Chronology of agricultural activities in the Canadian prairies	2.43
2.10 The relationship between food resources and critical periods of demand by laying hens and broods	2.51
3.1 Waterfowl strata utilized for annual survey purposes	3.1
3.2 Mean breeding duck densities averaged over the 1970-1983 period for strata 26 to 40	3.3
3.3 Distribution of the main crops grown in the prairie provinces in 1972	3.12
3.4 Map of the prairie provinces showing the overlap between census divisions and waterfowl strata	3.15
3.5 Flow chart of calculations used to apportion ducks to specific crop types	3.17
3.6 May and July pond numbers, total ducks and a producitivity index plotted as deviations from their 29-year mean (1955-1983) for strata 32 and 33 combined in SW Saskatchewan	3.18

FIGURE	FOLLOWING PAGE
5.1 Surface area estimated to have been sprayed aerially with pesticides in the three prairie provinces, 1957 to 1985	5.2
5.2 Contribution of runoff to total herbicide input to potholes	5.14
6.1 Hypothetical dissipation curve for a single application of a slightly soluble organic insecticide in water of a shallow pond	6.20
6.2 Schematic of the four-compartment model	6.21
6.3 Plot of growth of Black Duckand Mallard ducklings under optimal laboratory conditions and from uncontaminated and contaminated pond habitats	6.39
6.4 Average cumulative growth, dry matter intake and live weight change for captive Mallard ducklings during their first four weeks	6.41
6.5 Average cumulative growth, dry matter intake and live weight change for captive Lesser Scaup ducklings during their first four weeks	6.41
6.6 Plot of growth for Mallard ducklings based on the DUCKLING GROWTH MODEL and data from Sugden et al. (1981)	6.44
7.1 Hypothetical plot of build-up of herbicide residues in soil and sediment based on percent yearly carry-over of the undegraded portion of the compound	7.3
8.1 Possible points of impact of herbicides on ducks during their breeding cycle	8.3
8.2 Reductions in successful nest production of ducks under various scenarios of herbicide effects	8.7

LIST OF TABLES

TABLE	FOLLOWING PAGE
2.1 Climate data for three sites in Saskatchewan	2.2
2.2 Average number of potholes by size class	2.3
2.3 Pothole densities by stratum and vegetation zone	2.3
2.4 Wetland types	2.3
2.5 Densities of various types of wetlands summarized by vegetation zone	2.3
2.6 Average density of potholes at Redvers, Saskatchewan, during drought and non-drought years	2.4
2.7 Pothole water chemistry from various sites in the prairie pothole region	2.6
2.8 Ionic dominance patterns as a function of conductivity in 80 small prairie wetlands	2.7
2.9 Species of vegetation common in uplands and wetlands of the Canadian prairies	2.8
2.10 Distribution of breeding birds among classes of wetlands	2.9
2.11 Nutrient content of some spring runoff samples from five lake basins in the Erickson-Elphinstone district of southwestern Manitoba	2.16
2.12 Nitrogen and phosphorus concentrtions in three wetland compartments	2.17
2.13 Annual production and maximum standing crop values for some wetland macrophytes	2.19
2.14 Primary production of the major ecosystems of the world	2.19
2.15 Primary production of epiphytic algae on macrpophytes	2.22
2.16 Invertebrate taxa found in prairie sloughs	2.22
2.17 Invertebrate standing crop values for some ecosystems and habitats	2.24
2.18 Abundance of invertebrates associated with various aquatic plant species	2.26
2.19 Feeding mechanism and food types of aquatic insects important in potholes	2.31

TABLE	FOLLOWING PAGE
2.20 Feeding mechanisms and food types for invertebrate taxa important to potholes	2.31
2.21 Duck species with breeding populations of greater than 100,000 in the Canadian prairies	2.35
2.22 Density of breeding ducks on wetlands containing ponded water	2.38
2.23 Habitat preferences of breeding ducks in the prairie pothole region	2.38
2.24 Nesting habitat requirements summarized for duck species in the Canadian prairie regions	2.41
2.25 Summary of key studies relating nest density and nest success of dabbling ducks to crop type	2.43
2.26 Descriptions of brood habitat and brood mobility	2.44
2.27 Percentage of major types of foods consumed by hens of certain species of wild ducks during the laying period in the prairie pothole region	2.46
2.28 Invertebrate prey selected by hens of wild ducks during their reproductive period	2.46
2.29 Nutrient composition of certain plant and animal foods consumed by female dabbling ducks breeding in the prairie pothole region	2.46
2.30 Percentage of major foods consumed by ducklings prior to attaining flight in brood foraging areas in the prairie pothole region	2.49
2.31 Period of duckling growth during which invertebrates account for more than 70% of food consumed	2.50
2.32 Invertebrate prey selected by flightless ducklings foraging in the prairie pothole region	2.50
2.33 Plant foods selected by flightless ducklings foraging in the prairie pothole region	2.50
2.34 Indices of reproductive success for ducks	2.52
2.35 Nest density and nesting success for selected studies of communities of dabbling and diving ducks in specific habitats and under various environmental conditions in the prairie pothole region	2.53

TABLE FOLLOWING PAGE

2.36 Pair and nest density and nesting success for duck species from selected studies in the prairie pothole region 2.54

2.37 A summary of clutch size and percent hatching success data for duck species in the prairie pothole region 2.55

2.38 Measures of brood and recruitment success for duck species from selected long-term studies in the prairie pothole region 2.56

3.1 The vegetation form and surface area of waterfowl strata 3.1

3.2 Population estimates of ducks and American Coot in the Canadian prairies compared to the total surveyed population levels and to current Canadian and U.S. population levels of the same species 3.2

3.3 Comparison of breeding duck densities in the vegetation zones of each prairie province: 1960-69 versus 1970-83 3.2

3.4 CLI potential waterfowl habitat distribution by management district 3.6

3.5 Estimated mean duck densities for CLI waterfowl classes 3.7

3.6 Estimated mean duck numbers by CLI land class in the four prairie "duck management districts" 3.8

3.7 General types of wetland losses in midcontinent North America 3.8

3.8 Quantitative wetland use conversion studies in western Canada 3.9

3.9 Proportion of Agricultural Class 1-7 lands in cropland, by province 3.11

3.10 Agricultural land-use change in the prairies between 1951-1981 3.11

3.11 Agricultural land-use change in Alberta between 1951 and 1981 3.11

3.12 Agricultural land-use change in Saskatchewan between 1951 and 1981 3.11

3.13 Agricultural land-use change in Manitoba between 1951 and 1981 3.11

3.14 Area of crops on farms, 1981 3.12

TABLE FOLLOWING PAGE

3.15 Overlap between the stated Waterfowl Classes and Agriculture Classes 1-3 by region within the prairie (grassland/parkland) expressed as a percent of the total area within the designated Waterfowl Class(es) 3.14

3.16 Overlap between the stated Waterfowl Classes and Agriculture Classes 1-4 by region within the prairie (grassland/parkland) expressed as a percent of the total area within the designated Waterfowl Class(es) 3.14

3.17 Number of breeding ducks and % of total estimated prairie population within designated Waterfowl Classes which overlap with Agriculture Classes 1-3 3.14

3.18 Number of breeding ducks and % of total estimated prairie population within designated Waterfowl Classes which overlap with Agriculture Classes 1-4 3.14

3.19 Estimated number of ducks of all species breeding on land currently under crop in the grasslands-parklands region of the prairie provinces 3.15

3.20 Nest densities of ducks for various habitats 3.17

3.21 Major crop areas in the grassland-parkland region of the prairie provinces and estimated numbers of ducks nesting within or in association with these crops 3.17

3.22 Total number of breeding ducks and May and July ponds, total brood index and productivity index in Strata 32 and 33 combined for the period 1955-1983 3.18

4.1 Insecticides used for grasshopper control (1961-72) in Alberta 4.6

4.2 Information on the 1985 grasshopper spray program 4.6

4.3 Area sprayed with dimethoate for grasshopper control in Saskatchewan 4.6

4.4 Grasshopper forecasts for the prairie provinces 1973-1983 4.7

4.5 Areas of land sprayed for grasshopper control as a percentage of forecast from previous year 4.7

4.6 Summary of reported estimates of crop sprayed for pest control (other than grasshopper), 1970-1983, from Canadian Agricultural Insect Pest Review (Agriculture Canada) 4.8

TABLE	FOLLOWING PAGE
4.7 Estimated average area treated during the 1971-72 bertha armyworm infestation	4.8
4.8 Reported areas sprayed or dusted for insect control in the prairie provinces, 1962, 1970-82	4.8
4.9 Estimated area sprayed in 1983 for insect control based on the EPS registrant survey	4.9
4.10 Period of insecticide application for several common insect pests in the prairies	4.10
4.11 Percent of crop areas treated for two major weed categories in Manitoba and Saskatchewan between 1977 and 1979	4.13
4.12 Proportion of cropped hectares in Manitoba treated with wild oat herbicides, 1982	4.13
4.13 Estimated crop hectarage sprayed annually with broadleaf or grassy weed herbicides in the grassland-parkland area of the prairies as defined in Chapter 3	4.13
4.14 Percent of cereal crop area receiving different herbicides in different years	4.13
4.15 Market share of wild oat herbicides in Manitoba in 1982	4.14
4.16 Fungicides currently registered as seed treatments in oilseed, cereal and forage crops	4.16
4.17 Fertilizer use in western Canada for selected years between 1938 and 1979	4.17
5.1 Effect of drop size on terminal velocity, drift distance and coverage	5.4
5.2 Drop size distribution in aerosols and sprays	5.4
5.3 Off-swath deposit of aerially applied pesticides	5.10
5.4 Seasonal losses of pesticides in agricultural fields due to surface runoff	5.13
5.5 Herbicide runoff from agricultural fields under "worst case" conditions of high rainfall shortly after application	5.13
6.1 Summary of laboratory studies comparing the toxicities of synthetic pyrethroids and selected insecticides to larval dipterans	6.1

TABLE | FOLLOWING PAGE

6.2 Summary of toxicity data for synthetic pyrethroid effects on non-target aquatic insects 6.2

6.3 Summary of selected toxicity data for aquatic insects exposed to insecticides 6.3

6.4 Summary of toxicity data for synthetic pyrethroid effects on aquatic crustaceans and molluscs 6.3

6.5 Summary of selected toxicity data for cladocerans exposed to insecticides 6.4

6.6 Summary of selected toxicity data for amphipods exposed to insecticides 6.4

6.7 Selected reports of model ecosystem and field toxicity tests with synthetic pyrethroid insecticides 6.7

6.8 Selected reports of model ecosystem and field toxicity tests with carbamate, organochlorine and organophosphorous insecticides 6.11

6.9 Summary evaluation of proposed models of ecological hazard in aquatic ecosystems 6.18

6.10 Definition of the standard pond system 6.21

6.11 Constraints on input data and effects of input parameters on pollutant dynamics 6.21

6.12 Data from literature describing the physico-chemical properties and degradation contents of those insecticide compounds to be analysed in the aquatic persistence model 6.21

6.13 Toxicity test chronicity for Gammarus lacustris exposed to insecticides 6.25

6.14 Summary of toxicity values for integrated median lethal exposure model reported as range from available data sets 6.27

6.15 Application rates as input into the relative hazard model normalized when possible to control of grasshoppers on cereal crops 6.28

6.16 Computer model predicted distribution and degradation of selected synthetic pyrethroid insecticides in a simulated shallow pond 6.29

TABLE FOLLOWING PAGE

6.17 Computer model predicted distribution and degradation of selected major use agricultural insecticides in a simulated shallow pond 6.29

6.18 Relative risk index values for *Daphnia magna*, *Gammarus lacustris*, aquatic insects, *Chironomus* spp., *Culex* spp. or comparable taxa, based on the ACUTE EXPOSURE MODEL 6.34

6.19 List of insecticides reflecting relative hazard rankings based on ACUTE EXPOSURE MODEL predictions and short-term laboratory toxicity test data 6.35

6.20 Relative risk index values for *Daphnia magna*, *Gammarus lacustris*, aquatic insects, *Chironomus* spp., *Culex* spp. or comparable taxa, based on the CHRONIC EXPOSURE MODEL 6.35

6.21 List of insecticides reflecting relative hazard rankings based on CHRONIC EXPOSURE MODEL calculations and retentive capacity predictions 6.35

6.22 Summary of expected adverse response of aquatic invertebrate community in treated ponds for suggested benchmark insecticides 6.37

6.23 Predicted macroinvertebrate standing crop necessary to support various levels of growth rate for Mallard and Lesser Scaup ducklings 6.44

7.1 Persistence of select herbicides in Canadian agricultural soils 7.3

7.2 Physico-chemical properties of selected herbicides and formulations 7.5

7.3 Classes of compounds related to the effect of pH on adsorption of herbicides 7.8

7.4 Expected equilibrated distribution of selected herbicides in a contaminated slough ecosystem 7.10

7.5 Grouping of selected herbicides according to expected persistence in pond ecosystems 7.13

7.6 Persistence of selected herbicides under field conditions 7.13

7.7 Normal application rates and expected initial concentrations in pond water from aerial overspray of herbicides 7.14

TABLE	FOLLOWING PAGE
7.8 Herbicide concentrations tested which resulted in partial or total inhibition of growth or photosynthesis in various algal species	7.15
7.9 Effect of aerial application of three herbicides on aquatic vascular plants	7.17
7.10 Recommended application rates of six herbicides for control of aquatic crop weeds	7.17
7.11 Effects of three herbicides on emergent vegetation	7.18
7.12 Effects of seven herbicides on aquatic vascular plants in laboratory and field tests	7.18
7.13 Time scale of some events following application of herbicides for aquatic weed control	7.20
7.14 Recorded effects of herbicides on aquatic plants	7.22
7.15 Summary of selected toxicity data for herbicide effects on zooplankton	7.25
7.16 Summary of toxicity data for herbicide effects on aquatic amphipods, isopods and ostracods	7.25
7.17 Summary of toxicity data for herbicide effects on aquatic insects	7.25
7.18 Summary of toxicity data for herbicide effects on aquatic snails and worms	7.25
7.19 Ratio of 48 h LC50 values to calculated maximum expected initial concentrations for the more toxic herbicide compounds	7.26
7.20 Foods of ducks in western Canada	7.31
7.21 Predominant plant foods consumed by 11 species of waterfowl in the prairies	7.31
8.1 Parameter values entered in the calculation of successful nest production in the agricultural areas of the grassland-parkland region of the prairies	8.4
8.2 Nest success of ducks by habitat type	8.4
8.3 Number of successful nests produced, by land use and habitat type, in the grassland-parkland region under a "no herbicide effect" scenario	8.6
8.4 Hazard scenarios with insecticide use	8.10

TABLE FOLLOWING PAGE

8.5 Summary of ducks affected under three scenarios of insecticide use 8.13

PERSONAL COMMUNICATIONS

Adams, G.D.	Canadian Wildlife Service (C.W.S.), Saskatoon
Akesson, N.B.	University of California, Davis
Bidleman, T.	University of South Carolina
Brace, R.K.	C.W.S., Saskatoon
Burroughs, R.	Statistics Canada
Cooch, F.G.	C.W.S., Ottawa
Desgrange, J.-L.	C.W.S., Quebec
Dolinsky, M.	Alberta Agriculture, Edmonton
Driver, E.	C.W.S., Saskatoon
Ewen, A.	Agriculture Canada, Saskatoon
Greenwood, R.	USFWS Northern Prairie Wildlife Research Centre, Jamestown, North Dakota
Haramis, M.	USFWS Patuxent
Harris, L.	Saskatchewan Agriculture
Kaminski, R.M.	Ducks Unlimited, Winnipeg
King, G.J.	Agriculture Canada (formerly)
Kolack, A.	Manitoba Agriculture, Winnipeg
Krapu, G.L.	USFWS Northern Prairie Wildlife Research Centre, Jamestown, North Dakota
Laurenz, R.	Dept. of Natural Resources, Minnesota
Maybank, J.	Saskatchewan Research Council, Saskatoon
Millar, J.B.	C.W.S., Saskatoon
Nerassen, T.	Ducks Unlimited, Winnipeg
Nicholaichuk, W.	Inland Waters Directorate, Saskatoon
Philip, H.	Alberta Agriculture, Edmonton

Roberts, J.R.	NRCC, Ottawa (formerly)
Shaw, G.	C.W.S. Saskatoon
Smith, D.	Manitoba Agriculture Winnipeg
Swanson, G.	USFWS Northern Prairie Wildlife Research Centre, Jamestown, North Dakota
Todd, B.G.	Manitoba Agriculture, Winnipeg
Weaver, H.	Alberta Environment
Whitehead, P.E.	C.W.S., Vancouver

CHAPTER 1

INTRODUCTION

The pothole region of the western prairie is known as the bread-basket of Canada. Yet, this unique region is dotted with a vast number of freshwater ponds and lakes which are primarily small, shallow and biologically productive sloughs. These sloughs are critical to a wide variety of wildlife, the most notable of which are an estimated 16 million ducks that breed there annually. There is an intimate relationship between the ecology of the sloughs and the essential resources they provide to the waterfowl species that inhabit them for part of each year. The overlap between agriculture and waterfowl over much of this habitat has been a great concern to wildlife agencies for some years. Drainage programs to increase the quantity of arable land have resulted in the loss of valuable duck breeding habitat and agricultural practices such as the cultivating and burning of fields as well as their margins have contributed to the degradation of much of the remaining waterfowl habitat in agricultural regions. The potential impact of the agricultural use of pesticides on the reproductive success of Canadian duck populations has received only scant attention.

As pesticide manufacturers endeavour to expand the margin of safety between the median lethal response to target invertebrates and levels toxic to man and other vertebrates, concern over direct effects on birds should diminish (Appendix 1.1). However, indirect effects such as reduction or elimination of food or cover will continue. The extreme toxicity of the newer synthetic pyrethroid insecticides and the high toxicity of many other insecticides in current use in Canada to aquatic organisms raises the spectre of potentially massive invertebrate kills in contaminated slough systems. The request for the aerial application of one of the synthetic pyrethroids, deltamethrin, has provided the impetus to comprehensively evaluate the threat of broadscale insecticide-induced mortality within the sloughs' invertebrate communities, and the subsequent impact on duck reproductive success. The rationale for examining the indirect impacts of herbicides on waterfowl is also well founded. There are well-documented preferences of waterfowl for specific habitat and plant food resources. Herbicides have been shown to cause changes in vegetation which may affect the nesting success, predator avoidance and foraging of avian species. In addition, herbicides may either be highly toxic to aquatic invertebrates or affect their abundance via a reduction in the macrophyte substrate.

Because of the interdisciplinary nature of an ecotoxicological evaluation, this report deals with the use, environmental behaviour and toxic impacts of a large number of pesticides as well as the ecology of sloughs and the reproductive and feeding ecology of ducks in this type of habitat. The scope of the report is, therefore, quite broad. The theme of potential hazard of pesticides to waterfowl, however, provides a continuous thread.

An overview of the ecology of sloughs and of the duck species that use these valuable resources during their crucial reproductive phase is presented in Chapter 2. Initial estimates of the overlap between agricultural activity and waterfowl habitat are defined in Chapter 3. The quantitative use patterns of pesticides in the Canadian prairies and some of the inadequacies of present data collection are discussed in Chapter 4. A brief discussion of the contributions of ground and aerial application and agricultural runoff to the likelihood of pesticide entry into potholes is included in Chapter 5. The toxic effects of insecticides on aquatic invertebrates, with specific emphasis on the synthetic pyrethroid compounds, are evaluated from laboratory and field data as well as a hazard model in Chapter 6. Adverse resource modifications from herbicide use are evaluated using a combination of laboratory and field data on toxicity and persistence in Chapter 7. The focus is on the few compounds for which there is a substantial and relevant data base. The report is concluded in Chapter 8 with a quantitative assessment of insecticide and herbicide hazards to breeding duck populations in the Canadian prairies under expected use patterns. Assumptions and data gaps are clearly stated to accentuate the uncertainties inherent in this evaluation. An executive summary (chapter 9) provides the reader with highlights of the report and suggestions for further research.

Appendix 1.1. The toxicity of current prairie pesticides to waterfowl

The acute or chronic toxicity of modern pesticides to wildlife in general (Grue _et al_. 1983), and waterfowl in particular (Grue _et al_., in press), has been reviewed extensively. We chose, in this report, to concentrate instead on the ecologically-mediated impacts of pesticides because they have not received nearly as much attention. However, we cannot completely ignore the potential for the direct toxicological impact that exists in today's agricultural landscape and indeed, acute toxicity considerations are part of our final hazard assessment in Chapter 8. A brief summary of the acute dietary and reproductive toxicity of the main prairie pesticides is given below. For a review of the possible sublethal impacts, refer to two references cited above.

Laboratory Toxicity Data

Summaries of the direct toxicity of selected insecticides to the standard waterfowl test species, the Mallard Duck (_Anas platyrhynchos_), are presented in Tables A and B. Acute oral toxicity is measured as mg toxicant per kg weight of food consumed. Carbofuran has the lowest reported acute oral LD_{50} of the compounds listed, 0.397 mg kg^{-1} for 3-month-old ducklings. This value is a factor of 10 lower than the next more toxic compounds, diazinon and demeton. A number of other organophosphorous and carbamate compounds have oral LD_{50} values of less than 100 mg kg^{-1}. These include chlorpyrifos, dimethoate, methomyl and oxydemeton methyl. Azinphos methyl is slightly less toxic. The remaining compounds, including the synthetic pyrethroids, have LD_{50} values greater than 1000 and as such are relatively non-toxic to the Mallard under conditions of acute oral exposure. Dietary LC_{50} values are generally greater than 100 mg kg^{-1} food with the exceptions of carbofuran, chlorpyrifos, demeton and diazinon.

There are few data on the impacts of multiple low-level exposures to a single insecticide on avian reproduction. The scant data available have been generated on a number of avian species and often not according to currently accepted protocols. Due to small sample sizes, only dramatic impacts can be seen, and biochemical or histological investigations are rarely carried out. The data indicate that diazinon, dimethoate, malathion and methomyl have embryotoxic and teratogenic effects (Table C), and only a slim margin of safety is reported for malathion at its recommended application level.

Summaries of available published laboratory data on the toxicity of herbicides to Mallard Ducks are presented in Tables D and E. The data sets are incomplete, but reported lethal concentrations are relatively high. Only metribuzin, paraquat and bromoxynil are acutely toxic to orally exposed Mallards at levels less than 200 mg kg^{-1} bird weight and only linuron was toxic to ducklings via dietary intake at levels less than 5000 mg kg^{-1} food. The LD_{50} values for these more toxic herbicides are three orders of magnitude greater than LD_{50} values for the more toxic insecticides. Recent tests for herbicide effects on Mallard embryos indicate that trifluralin, paraquat, propanil, diclofop-methyl and bromoxynil (with MCPA) are application levels (Table C). Thus, at least for these compounds, the possibility of direct toxic impacts to waterfowl reproductive success should not be overlooked. The prediction of direct toxic effects is complicated by the fact that certain herbicides contain contaminants which are highly toxic (e.g. TCDD in 2,4,5-T).

There are almost no data on effects from simultaneous or sequential exposures to a mixture of toxic compounds or even from formulated products or final tank mixes as applied in the field.

Field Evidence of Direct Toxicity

Some of the insecticides in common use in the prairies have been implicated in kills of waterfowl when apparently used according to label instructions. For example, between 1972 and 1977, at least six cases of mass mortality of geese and dabbling ducks totalling over 5,000 birds were reported following the use of a liquid formulation of carbofuran (Furadan 4F) on alfalfa. Over the same period, approximately 1,400 dead dabbling ducks were picked up in five separate incidents involving granular carbofuran (Furadan 10G) in turnip and potato fields and a liquid formulation of carbofuran (Furadan 4.8F) in potato fields (NRCC 1979, Flickinger et al. 1980, Hill & Fleming 1982). Kills of species other than waterfowl have also been reported with this pesticide.

Diazinon has also been associated with waterfowl mortality. Between 1974 and 1980, a large number of waterfowl (primarily goose) kills on turf treated with liquid or granular formulations of diazinon were reported (Schobert 1974, Zinkl et al. 1978, Stone 1979, 1980, Stone & Knoch 1982, Stone & Gradoni 1985, USEPA 1985). These kills ranged from a few to 1,000 birds per event. Hill and Fleming (1982) reported a kill of 140 geese and dabbling ducks on or adjacent to a field (crop type unspecified) which had been sprayed aerially with diazinon the preceeding afternoon. In Canada diazion is not used on field crops proper but rather it is used to create "barrier strips" against grasshoppers along field margins.

Chlorpyrifos has been implicated in one case of mortality on turf (Stone 1979) and Hurlbert et al. (1970) documented a higher mortality rate for ducklings confined to ponds treated with four dosages of chlorpyrifos than for controls. Since mortality in the latter study occurred soon after treatment and was not dose-dependent, and since ducklings were seen ingesting incapacitated macroinvertebrates, the authors speculated that the ducklings were poisoned through ingestion of contaminated prey.

Methamidophos has been implicatead as the toxic metabolite in the prolonged intoxication and brain cholinesterase inhibition of songbirds in forest plots sprayed with acephate (o,s-dimethyl acetylphosphoramidothioate) (Julin & Gramlich 1978, Zinkl et al. 1979, 1980, 1981). However, no waterfowl impacts have been reported for this compound.

Clearly, the direct hazard to waterfowl of these highly toxic insecticides is a function of their toxicity, the extent of their use and the available routes of exposure. Carbofuran and diazinon are the two most acutely toxic of these insecticides and this is reflected in the high number of kills reported. Grue et al. (1983) showed that a large proportion of reported bird kills attributed to organophosphates could be predicted on the basis of toxicity and use statistics. Granular insecticides create a special hazard to birds since granules tend to be selectively picked up as grit or seed when they are left on the soil surface. Current machinery in use does not ensure complete incorporation of these granules (Balcomb et al. 1984). For ducklings or smaller songbirds, one granule of the more toxic insecticides (e.g. carbofuran) often exceeds a lethal dose and calculations show that an LD_{50} for an adult dabbling duck is in the range of 2-3 granules of carbofuran.

As pointed out by Banks (1979), it is impossible at present to assess the true magnitude of direct avian kills resulting from pesticide use. Although we might expect waterfowl kills to be reported proportionately more often than kills of other species in view of the relatively large body size of waterfowl, a number of factors mitigate this reporting rate: (1) Fields are seldom, if ever, checked for the presence of kills. Most of the kills cited above occurred on or near waterfowl refuges (e.g. Hill & Fleming 1982) or were part of a directed study stemming from a personal interest of the researcher for turf insecticides (e.g. Stone 1979, 1980, Stone & Knoch 1982, Stone & Gradoni 1985). (2) Carcasses of birds can rapidly disappear through the combined effects of weathering and scavenging (Rosene & Lay 1963) or through human interference. Stone (1980) described a case where goose carcasses from a diazinon kill were being picked up and buried as an investigator arrived on the scene. (3) There is undoubtedly a reticence on the part of the applicator to report any kills since this results in an investigation by local wildlife authorities. A recent attempt was made by CWS to elicit the help of growers by distributing, in person, a questionnaire on wildlife kills coinciding with a large-scale efficacy trial of a new insecticide. Approximately half of the 100 growers contacted were reportedly hostile to the notion of this survey and none of the questionnaires were mailed back to CWS even though stamped addressed envelopes had been supplied. Only eight responses were eventually elicited by the registrant's field staff in the course of their follow-up visits. Similarly, a toll-free bird kill "hot-line" in British Columbia has not turned up any cases since it was instituted in 1982 (Whitehead, pers. comm.). Clearly, the only way to gain meaningful information on the importance of direct toxic impacts is through research by or under the direct supervision of interested agencies (e.g. Balcomb et al. 1984).

Appendix 1.1.

Table A. Acute oral[a] toxicity of selected insecticides to Mallard Ducks (*Anas platyrhynchos*).

Insecticide	Sex	Age	LD_{50} mg kg^{-1} Body wt (95% CL)	Number of LD_{50}-equivalent doses applied per m^2 at maximum application rates[b] (mg ai kg^{-1} ha^{-1})	References
azinphos methyl	male	3-4 m	136 (97.8-188)	.74	Tucker & Crabtree 1970
carbaryl	female	3 m	> 2179	< 10	Tucker & Crabtree 1970
	female	12 m	> 2564	< 09	Hudson *et al*. 1984
carbofuran	female	3 m	0.397 (0.315-0.500)	69	Tucker & Crabtree 1970
	female	12 m	0.510 (0.410-0.635)	54	Hudson *et al*. 1984
	male	12 m	0.480 (0.381-0.604)	57	Hill *et al*. 1975
chlorpyrifos	female	--[c]	75.6 (35.4-161)	.73	Tucker & Crabtree 1970
	male	--	70-80	.69	"
	male & female	17 d	167 (78.3-357)	.33	"
	male & female	15-19 d	112 (11.5-1089)	.49	Hudson *et al*. 1984
demeton	male	3 m	7.19 (5.19-9.97)	4.5	Tucker & Crabtree 1970
diazinon	male	3-4 m	3.54 (2.37-5.27)	d	Tucker & Crabtree 1970
dimethoate	male	3-4 m	41.7 (30.1-57.8)	1.3	Tucker & Crabtree 1970
	female	3-4 m	63.5 (45.8- 88.1)	.87	Hudson *et al*. 1984
malathion	female	3-4 m	1487 (1020-2150)	.06	Tucker & Crabtree 1970
methamidophos	male	3 m	8.48 (6.73-10.7)	6.6	Hudson *et al*. 1984
methidathion	female	3-4 m	23.6 (16.5-33.8)	2.3	Hudson *et al*. 1984

Appendix 1.1.

Table A (cont'd)

Insecticide	Sex	Age	LD_{50} mg kg^{-1} Body wt (95% CL)	Number of LD_{50}-equivalent doses applied per m^2 at maximum application rates[b] (mg ai kg^{-1} ha^{-1})	References
methomyl	male	8 m	15.9 (12.6-20.0)	3.1	Tucker & Crabtree 1970
methoxychlor	male	3-4 m	<2000	<.14	Tucker & Crabtree 1970
naled	male	--	52.2 (37.8-72.3)	d	Tucker & Crabtree 1970
oxydemeton methyl	male	4 m	53.9 (38.9-74.8)	d	Tucker & Crabtree 1970
phosmet	male	3-4 m	1830 (1270-2630)	.06	Tucker & Crabtree 1970
trichlorfon	male	3-4 m	36.8 (26.6-51.1)	3.0	Hudson _et_ _al_. 1984
Synthetic Pyrethroids					
deltamethrin	male & female	14 d	> 4640	<.01	Elliott _et_ _al_. 1978
permethrin	male & female	--	> 13500	<.01	Elliott _et_ _al_. 1978

[a] Oral administration by inserting gelatin capsule containing insecticide through tube to level of crop or proventriculus

[b] Application rates are given in Appendix 4.1. Preference is given to rates registered against major field crop pests such as grasshoppers, bertha armyworm, etc.
This ratio is given in mg ai/kg/m^2 and is intended to provide a comparison between insecticides after taking into account differences in application rates.

[c] Age unknown

[d] No registration for major field-crop pest

Appendix 1.1.

Table B. Toxicity values for Mallard (*Anas platyrhynchos*) ducklings[a] exposed to selected insecticides via dietary intake[b].

Insecticide	Age	LC_{50} mg kg^{-1} food (95% CL)
azinphos methyl	10 d	1940 (978-4506)
carbaryl	24 d	>5000 [no mortality to 5000]
carbofuran	10 d	190 (156-230)
chlorpyrifos	10 d	940
demeton	10 d	598 (488-733)
diazinon	10 d	191 (138-253)
malathion	16 d	>5000 [no mortality to 5000]
methomyl	10 d	2883 (2000-4572)
methoxychlor	16 d	>5000 [no mortality to 5000]
naled	10 d	2724 (1068-15089)
oxydemeton methyl	10 d	>5000 [38% mortality at 5000]
trichlorfon	10 d	>5000 [30% mortality at 5000]

[a] Sex unknown

[b] Data on dietary effects of insecticides from Hill *et al.* 1975; dietary test based on 5 days of exposure via food followed by 3 days of observation

Appendix 1.1.

Table C. Effects of insecticides and herbicides in aqueous emulsion on Mallard (Anas platyrhynchos) embryos[a], when used at field crop levels[b].

Pesticide	LC_{50} in kg/ha (slope) [95% confidence limits]	LC_{50}/Maximum Rate of Application (kg/ha)	Reduced Growth[c]		Abnormal Survivors[d]		Malformations in Survivors (day 18)
			doses less than LC_{50}	doses equal to or greater than LC_{50}	doses less than LC_{50}	doses equal to or greater than LC_{50}	
Insecticides							
Carbaryl	>E.C.[e] = 242	>101	-	n.a.[f]	-	n.a.	g
Diazinon	83(3.80)[63-109]	151	+	+	-	+	incomplete caudal ossification, stunted
Dimethoate	94(1.77)[63-141]	196	+	+	+	+	bill, brain, gastroschisis, joint, stunted
Malathion	132(5.20)[104-167]	88	-	n.a.	-	+	bill, limb, skeletal scoliosis, stunted
Methomyl	>E.C. = 202	>390	+	+	+	n.a.	blister, edema, neck, stunted
Parathion	46(4.74)[34-63]	82	+	+	+	+	blister, edema, neck, skeletal scoliosis, lordosis, stunted
Permethrin	>45	>300	-	n.a.	-	n.a.	g
Phosmet	>561	>499	-	n.a.	-	n.a.	g

Appendix 1.1.
Table C (cont'd)

Pesticide	LC_{50} in kg/ha (slope) [95% confidence limits]	LC_{50}/Maximum Rate of Application (kg/ha)	Reduced Growth[c]		Abnormal Survivors[d]		Malformations in Survivors (day 18)
			doses less than LC_{50}	doses equal to or greater than LC_{50}	doses less than LC_{50}	doses equal to or greater than LC_{50}	
Herbicides							
Amitrole	197(3.47)[157-248]	39	+	+	-	+	stunted
Bromoxynil (with MCPA)	8.4(1.53)	21	-	+	-	+	edema, eye, stunted
2,4-D (dimethylamine salt)	215(2.10)[155-299]	168	-	+	-	-	g
Dalapon	>420	>334	-	n.a.	-	n.a.	g
Dicamba	>224	>1600	+	n.a.	+	n.a.	eye, stunted
Glyphosate	200(0.78)[117-341]	47	-	-	-	-	g
Diclofop-methyl	10.7(0.88)[5.3-20.3]	13	+	+	-	+	eye
Paraquat	1.7(1.66)[1.0-2.8]	1.5	+	+	-	+	brain, edema, stunted
Picloram	112(0.51)[84-149]	104	-	+	-	+	stunted
Propanil	8.1(1.49)[4.2-15.8]	7.9	-	+	+	+	limb, neck
Trifluralin	1.8(0.50)[1.0-3.1]	1.1	-	+	-	+	bill, stunted

[a] Modified from Hoffman and Albers 1984

[b] Maximum rate recommended for control in field crops in Canada. Rates are often higher in pastures, fencerows, shelterbelts, orchards etc. Phosmet and diazinon are not recommended for field crops; rates refer to maximum recommended for pasture. Maximum rate for parathion in forage used; rate is higher in hops and corn. Glyphosate is a non-selective herbicide.

[c] When differences between weights or crown-rump lengths of treated and control embryos were significant by one-way ANOVA (p 0.01) and Duncan's multiple range test (p 0.05), it is indicated by a "+". Non-significant differences are indicated by a "-".

[d] When the proportion of abnormal survivors from treated eggs was greater than for controls by Chi-square analysis (p 0.05), it is indicated by a "+". Non-significant differences are indicated by a "-".

[e] Greater than the emulsifiable concentrate (E.C.)

[f] Not applicable

[g] No malformations noted

Appendix 1.1.

Table D. Acute oral[a] toxicity of selected herbicides to Mallard Ducks (*Anas platyrhynchos*).

Herbicide	Sex	Age	LD_{50} mg kg^{-1} body wt (95% CL)	References
amitrole	male	3-4 m	>2000	Tucker & Crabtree 1970
asulam	--[b]	--	>4000	Weed Society of America 1983
bentazone	--	--	2000	Weed Society of America 1983
bromoxynil	--	--	200	Weed Society of America 1983
chloramben	female	3 m	1500 [pheasant]	Hudson *et al*. 1984
cyanazine	female	3-5 m	>2400	Hudson *et al*. 1984
2,4-D (acid)	male & female	3-5 m	>>1000	Tucker & Crabtree 1970
(sodium salt)	male & female	3-5 m	>>2025	"
(amine)	male	7 m	≈ 2000	"
(acid)	male	4 m	>2000	Hudson *et al*. 1984
dicamba	--	--	800 [pheasant]	Weed Society of America 1983
diquat	male	3-4 m	564 (324-982)	Tucker & Crabtree 1970
flamprop-methyl	--	--	>1000	Weed Society of America 1983
metribuzin	--	--	100	Weed Society of America 1983
paraquat	male	3-4 m	199 (144-276)	Hudson et al. 1984
picloram	male	3-4 m	>2000	Tucker & Crabtree 1970
trifluralin	female	3-4 m	>2000	Tucker & Crabtree 1970

[a] Oral administration by inserting gelatin capsule containing herbicide through tube to level of crop or proventriculus

[b] Sex or Age Unknown

Appendix 1.1.

Table E. Toxicity values for Mallard (Anas platyrhynchos) ducklings exposed to selected herbicides via dietary intake[ab].

Herbicide	Sex	Age	LC_{50} mg kg^{-1} food (95% CL)
amitrole	male	10 d	> 5000 [no mortality to 5000]
2,4-D (butoxyethmol ester)		23 d	> 5000 [no mortality to 5000]
(dimethylamine)	--[c]	17 d	> 5000 [no mortality to 5000]
dalapon	--	10 d	> 5000 [no mortality to 5000]
diclofop-methyl	--	--	20,000
diquat	--	16 d	>5000 [33% mortality at 5000]
flamprop-methyl	--	--	20,000
glyphosate	--	--	>4640
linuron	--	5 d	3083 (2419-3990)
MCPB	--	10 d	>5000 [no mortality to 5000]
paraquat	--	10 d	4048 (3432-4886)
picloram	--	10 d	>5000 [no mortality to 5000]
simazine	--	10 d	>5000 [no mortality to 5000] 51,200
terbacil	--	--	56,000 [Peking duckling]
triallate	--	--	> 5000

---

[a] Dietary test based on 5 days of exposure via food followed by 3 days of observation

[b] From Hill *et al*. 1975

[c] Sex or age unknown

CHAPTER 2

WATERFOWL AND PRAIRIE SLOUGH ECOLOGY

Assessing the hazards posed by pesticide use to waterfowl requires a full understanding of the habitat and food requirements of ducks and of the structure and functioning of the ecosystem. Contaminants can affect waterfowl through a number of different pathways from the moment of application. Determining the most likely route of contamination requires a knowledge of the physical environment in the prairie pothole region; understanding the soil and water chemistry will provide information on fate, persistence and toxicity of the compounds. The identification of the critical periods in the life cycle of ducks during their stay in the prairies requires knowledge about their use of the environment. In this chapter these various facets of duck and pothole ecology will be examined.

2.1 Physical Aspects of the Prairie Pothole Region

The prairie pothole region of North America covers about 300,000 square miles (770,000 km^2) in south-central Canada and the north-central United States (Figure 2.1). This swath, 300 miles (483 km) wide and 1,000 miles (1,609 km) long, crosses the provinces of Alberta, Saskatchewan and Manitoba and parts of the states of Minnesota, North and South Dakota, Iowa and Montana. Approximately two-thirds of this region is located in Canada with 64,000 square miles in Alberta, 114,000 in Saskatchewan and 39,000 in Manitoba (165,750 km^2, 295,250 km^2 and 101,000 km^2, respectively) (Gollop 1965).

2.1.1 Geology and soils of the prairie pothole region

Both ground and end moraines of undulating glacial till, now in various stages of erosion, characterize the topography of the prairie pothole region where the continental ice sheet advanced and retreated during the last ice age of the Pleistocene. The millions of shallow depressions and the lack of drainage characteristic of this landscape are the sources of the high concentration of potholes. The surface deposits consist of undifferentiated glacial till of shale, limestone and granitic rock. Delta plains and beds of glacial lakes resulting from deglaciation are found interspersed within the pothole region and consist mostly of silt, sand and clay deposits (Bird 1961, Weir 1971).

The prairie pothole region contains three major soil types: the brown, dark brown and black soils. The brown soils, found in southeastern Alberta and southwestern Saskatchewan, are low in nitrogen and develop under grassland vegetation in more arid conditions (Figure 2.2). The soils of the rest of the pothole region are dark brown to black and are formed under more productive grassland vegetation where average temperatures are lower and humidity is higher (Coupland 1961, Weir 1971).

Figure 2.1. Prairie pothole region of central North America (from Kantrud and Stewart 1977)

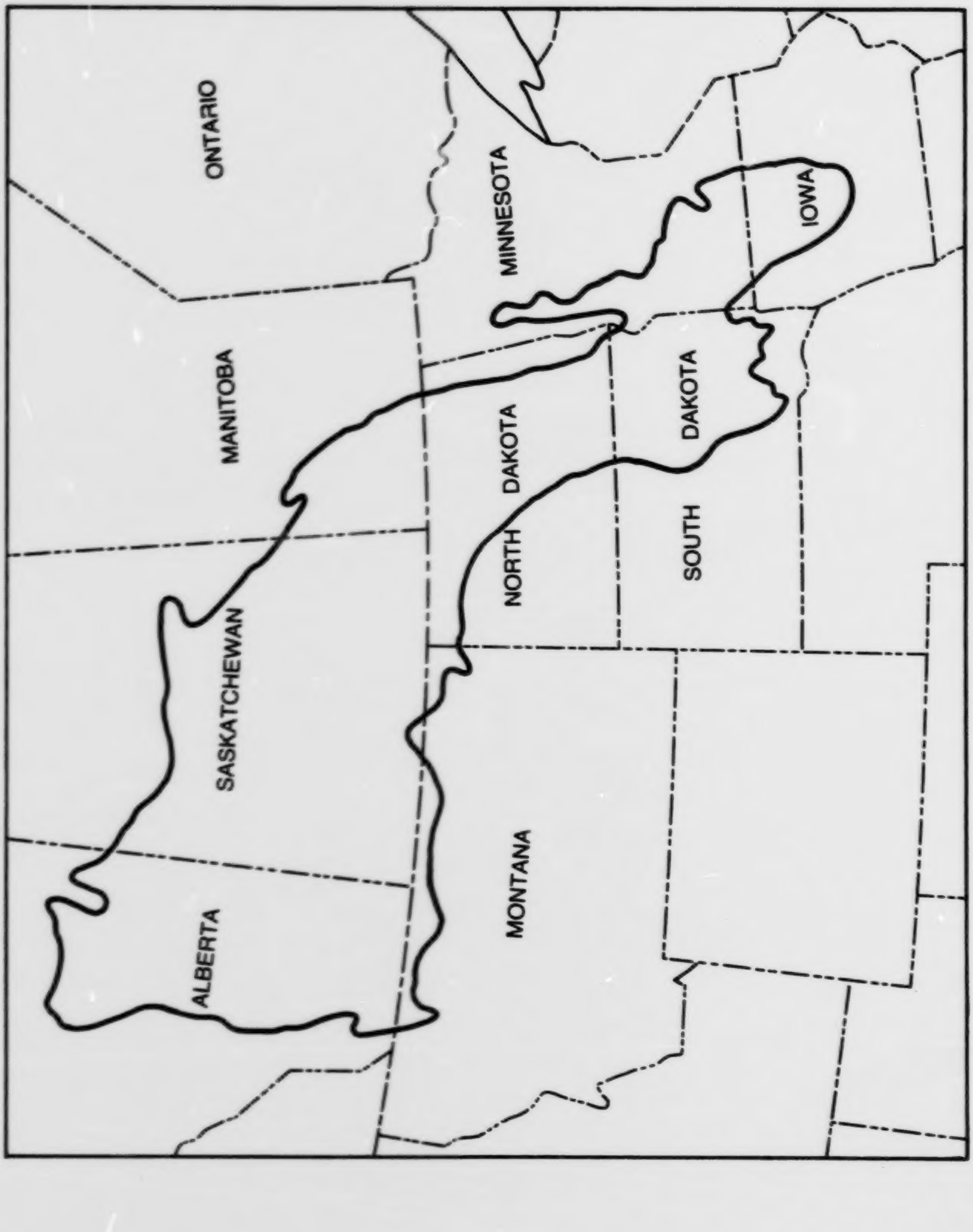
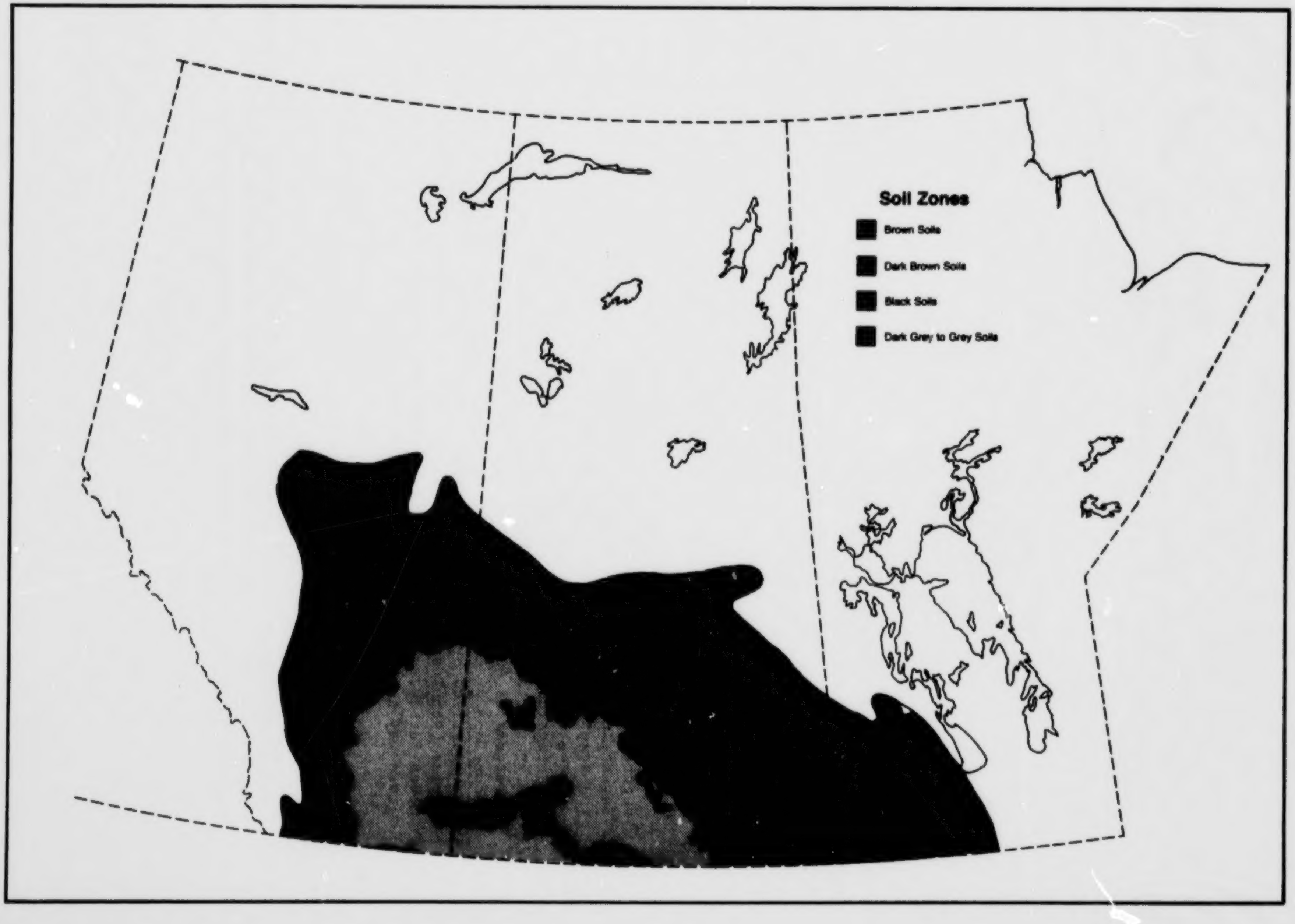

ONTARIO
MANITOBA
SASKATCHEWAN
ALBERTA
MINNESOTA
NORTH DAKOTA
SOUTH DAKOTA
IOWA
MONTANA

Figure 2.2. Soil zones of the prairie pothole region of Canada (from Weir 1971).


Soil Zones
Brown Soils
Dark Brown Soils
Black Soils
Dark Grey to Grey Soils

The texture of the soil is a function of the extent of weathering of the glacial deposits. Fine textured soils are found in areas of lacustrine deposits or where glacial drift has not been modified by water. Coarse textured soils are characteristic of glacio-fluvial deposits.

2.1.2 Climate

The climate of the prairie pothole region is semi-arid by Canadian standards. Average daily temperature ranges from -12 to -18°C in January and 17 to 20°C in July (Energy, Mines and Resources 1974). The mean annual precipitation is in the order of 30 to 46 cm and the average annual potential evapotranspiration is between 50 and 60 cm. Low levels of precipitation, combined with high evaporative losses due to the dry air masses coming down from the Rocky Mountains, are responsible for the aridity of the climate and the regular occurrence of droughts. Conditions are not homogeneous throughout the pothole region: as one moves from the grassland in the southwest to the parklands to the north and east the climate becomes less arid. Table 2.1 shows how wind speed and evaporation decrease as one moves northeast from Swift Current to Melfort while precipitation does not change significantly. This underlines the importance of the wind in creating drier conditions to the south.

2.1.3 Major vegetation zones

The climate of the prairies has a dominant influence on the vegetation. In drier areas such as southern Alberta and southwestern Saskatchewan, the grasslands dominate while aspen parkland constitutes the vegetstion of the northern and eastern part of the pothole region where water availability is not as limiting. Figure 2.3 illustrates the distribution of the two grassland zones and the aspen parkland zone. The shortgrass and mixed prairie (or medium grass) are dominated by wheatgrasses (*Agropyron* spp.), needlegrasses (*Stipa* spp.), grama grass (*Bouteloua gracilis*) and bluegrasses (*Poa* spp.). The aspen parkland is characterized by stands of trembling aspen (*Populus tremuloides*) and balsam poplar (*Populus balsamifera*) interspersed with grassland vegetation. As one moves further north, the forest becomes more prevalent as the aspen parkland gives way to the boreal forest (Bird 1961).

2.1.4 Pothole distribution

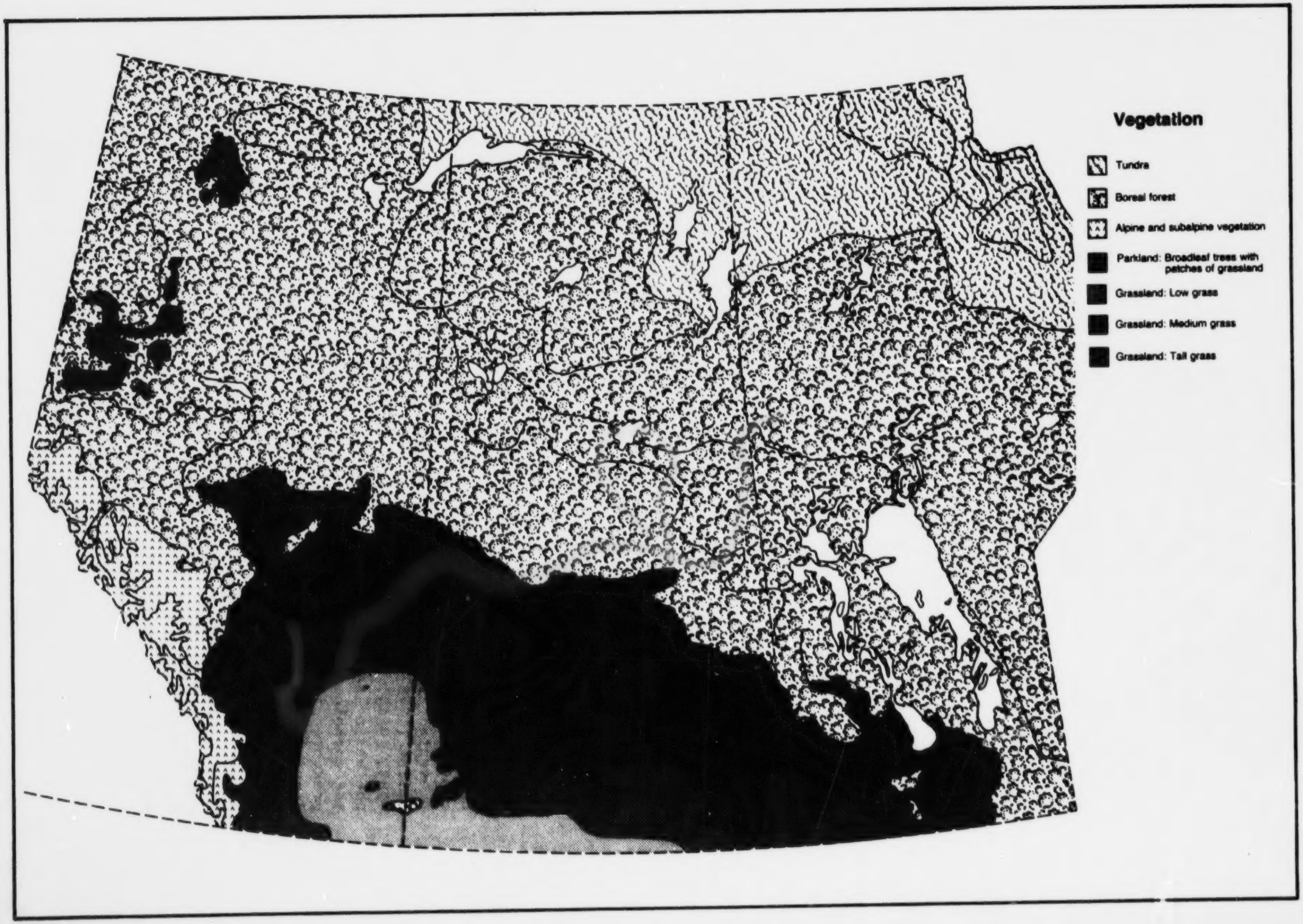
In the following discussion the words "pothole" and "slough" are used interchangeably and refer to any depression holding water. The term "basin", however, is defined as those depressions which are capable of holding water as indicated by the vegetation present (Millar 1969).

Table 2.1. Climate data for three sites in Saskatchewan[a].

Location	Average Annual Precipitation (cm)	Average Annual Wind Speed (km/h)	Average Annual Water Loss by Evaporation (cm)[b]
Swift Current	37.8	23.5	65.8
Saskatoon	35.9	17.5	54.9
Melfort	40.2	9.2	50.0

[a] Taken from Millar 1969
[b] Calculated from small dugouts

Figure 2.3. Major vegetation zones of the prairie provinces of Canada (from Weir 1971).


Vegetation
Tundra
Boreal forest
Alpine and subalpine vegetation
Parkland: Broadleaf trees with patches of grassland
Grassland: Low grass
Grassland: Medium grass
Grassland: Tall grass

As the continental ice sheet receded, the sand and gravel basins were sealed with silt, allowing the retention of spring runoff waters which created a suitable environment for the growth of aquatic species. The average pothole density in the Canadian prairies is $12/km^2$, with values ranging from 4 to $40/km^2$ (Smith et al. 1964). In a wetland habitat study in Saskatchewan, Millar (1969) found basin densities to range between $31.4/km^2$ and $36.4/km^2$, occupying 8 to 12.5% of the total land surface; 82.0 to 87.5% of the basins were smaller than 1 acre (.4 ha) in size (Figure 2.4). The majority of potholes censused at Lousana (Alberta) and at Redvers (Saskatchewan) were smaller than 0.5 acres (.2 ha) in area (Table 2.2).

The distribution of potholes within the prairies is uneven. The shortgrass prairie has the lowest pothole density with an average of 1.6 $sloughs/km^2$ (Table 2.3). The mixed prairie and the more hilly parklands have densities three and almost five times, respectively, those of the shortgrass prairie. Even within the major vegetation zones, pothole numbers vary substantially: in the parklands, densities range from 2.9 to 14.2 $ponds/km^2$ for the various strata which are the physiographic units used in the annual aerial surveys of waterfowl breeding grounds run by the U.S. Fish and Wildlife Service (see also Section 3.2).

It should be noted at this point that the assignment of individual strata to specific vegetation zones may be misleading as most strata are not composed of a single vegetation type. For a more refined analysis of pothole distribution by dominant vegetation zone, one would need to go back to the raw data of the annual aerial surveys and divide strata into specific vegetation types. However, strata grouped to cover broad areas usually referred to as parkland, mixed prairie and shortgrass prairie, which are more than vegetation zones but usually incorporate landforms in their broad definitions (see Figures 2.3, 3.1 and Table 2.3), provide an adequate picture of general trends.

The pothole densities given above are most probably underestimates of actual pothole numbers because they were determined from airplanes during aerial surveys. Millar (1982, 1983, 1984) conducted field surveys of all depressions capable of holding water on 65 transects within the strata mentioned above. Each basin was classified according to the scheme presented in Table 2.4. This wetland classification scheme, developed by Millar (1976) and later expanded to include greater detail (Millar 1982, 1983, 1984), is based on the type of vegetation found in the deepest part of the basin (see Section 2.3.1) which reflects the permanency of water within the pothole. The data for 13 strata grouped into three vegetation zones is presented in Table 2.5. While again the parklands have the highest pothole densities followed by the mixed prairie and shortgrass prairie, the densities are much higher than those reported from the aerial surveys (Table 2.3). This is probably the result of the inclusion in Millar's survey of pothole types 0.1 to 1.9 which account for 57% of all potholes. These potholes are probably missed or underreported in aerial pond counts due to their shallow nature and the fact that they are often cultivated (type 0.3). Again, as in Table 2.3, Millar found a large variance in densities within vegetation zones.

Figure 2.4. Basin size distribution in three study areas of Saskatchewan.

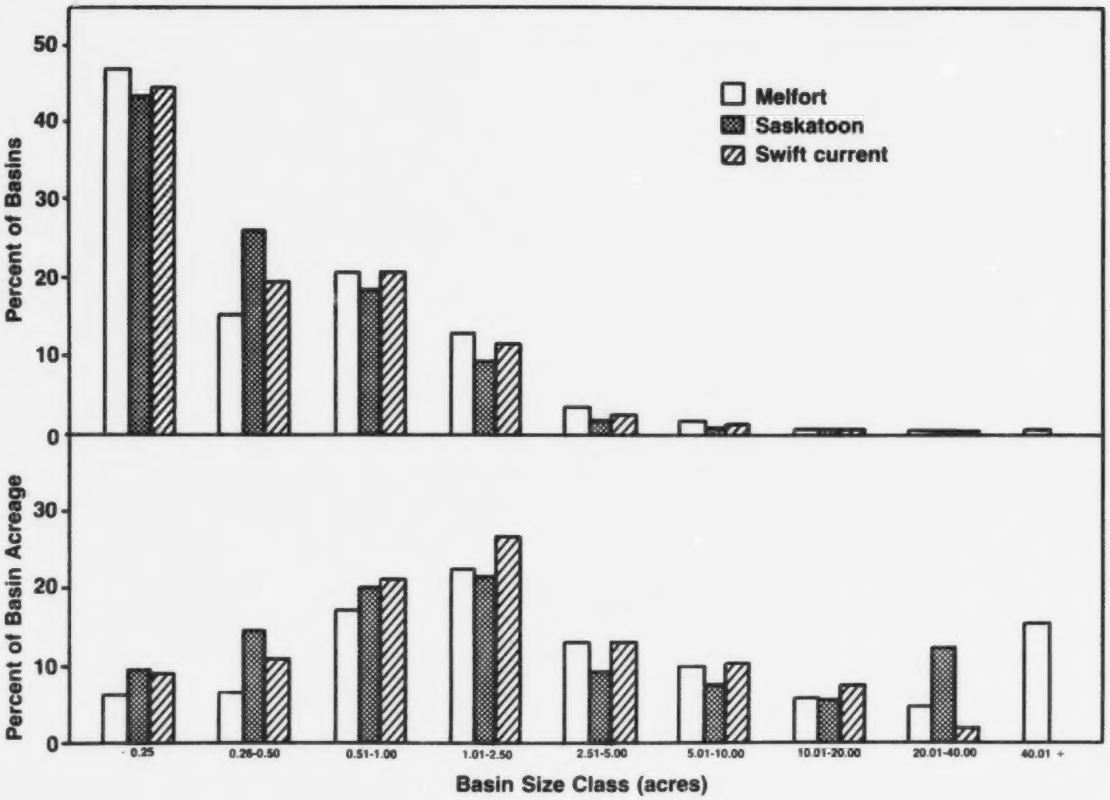
50
40
30
20
10
0
Percent of Basins
☐ Melfort
▩ Saskatoon
▨ Swift current
30
20
10
0
Percent of Basin Acreage
- 0.25
0.26-0.50
0.51-1.00
1.01-2.50
2.51-5.00
5.01-10.00
10.01-20.00
20.01-40.00
40.01 +
Basin Size Class (acres)

Table 2.2. Average number of potholes by size class for the period 1953-1965 at Lousana, Alberta (study area = 9.39 km^2), and Redvers, Saskatchewan (study area = 12.95 km^2)[a].

Size Class (acres)	Lousana		Redvers	
	May 1	July 1	May 1	July 1
0.0- 0.5	120	93	234	154
0.6- 1.0	18	16	32	24
1.1- 2.0	17	15	27	22
2.1- 5.0	13	13	13	10
5.1-10.0	0	0	2	2
20.0	1	1	0	0
Total	168	138	307	211

[a] Modified from Smith and Stoudt 1968

Table 2.3. Pothole densities by stratum and vegetation zone (average of 1955 to 1977)[a].

Vegetation Zone	Stratum	No. Ponds/km^2
Parklands	26	6.4
	30	5.0
	31	8.1
	34	14.2
	38	2.9
	40	13.9
		Average = 7.8
Mixed prairie	27	4.5
	28	2.9
	32	4.7
	35	12.9
	39	9.7
	45	4.9
	46	3.5
	48	2.3
		Average = 4.8
Shortgrass prairie	29	2.5
	33	3.2
	41	1.3
	42	1.0
	43	1.8
	44	1.4
		Average = 1.6

[a] Modified from Bellrose (1979)

Table 2.4. Wetland Types from Millar (1982, 1983, 1984).

Code No.	Wetland or Depression Type
0.0	Totally drained or filled depression or wetland
0.1	Herbaceous or shrub-filled depression - Low Prairie or drier species
0.2	Treed depression - aspen or other species
0.3	Unclassified depression - most commonly applied to cultivated sheetwater depressions in cropland
0.8	Low Prairie depression in abandoned channel or watercourse (oxbow)
0.9	Low Prairie depression in active, intermittent watercourse
1.1	Wet Meadow - herbaceous vegetation
1.2	Wet Meadow - willow-filled (shrub carr)
1.8	Wet Meadow - in abandoned channel or watercourse (oxbow)
1.9	Wet Meadow - in active, intermittent "beaded" watercourse
2.1	Shallow Marsh
2.8	Shallow Marsh - in abandoned channel or watercourse (oxbow)
2.9	Shallow Marsh - in active, intermittent "beaded" watercourse
3.1	Emergent Deep Marsh
3.8	Emergent Deep Marsh - in abandoned channel or watercourse (oxbow)
3.9	Emergent Deep Marsh - in active, intermittent "beaded" watercourse
4.1	Transitional Open Water Wetland
4.8	Transitional Open Water Wetland - in abandoned channel or watercourse (oxbow)
4.9	Transitional Open Water Wetland - in active, intermittent "beaded" watercourse
5.1	Open Water Marsh (<75% of wetland diameter)

Table 2.4 (cont'd)

Code No.	Wetland or Depression Type
5.2	Shallow Open Water Wetland (<75% of wetland diameter)
5.3	Artificial Shallow Open Water - a deepened natural wetland in which the excavation extends over 10% of wetland diameter
5.4	Artificial Shallow Open Water - totally man-made excavation (dugout, ditch, etc.) in an upland location
5.5	Artificial Shallow Open Water - shallow man-made dam or impoundment in an upland site
5.6	Artificial Shallow Open Water - man-made excavation or dugout in a watercourse or stream channel
5.7	Artificial Shallow Open Water - man-made or beaver dam or impoundment in a watercourse or stream channel
5.8	Shallow Open Water Wetland - in abandoned channel or watercourse (oxbow)
5.9	Shallow Open Water Wetland - in active, intermittent "beaded" watercourse
6.1	Open Alkali Wetland
8.1	Spring
8.2	Fen
8.3	Bog
8.4	Swamp
9.1	Artificial Stream Channel (drainage or irrigation ditch or stream diversion channel) - dominated by emergent (usually Deep Marsh) wetland vegetation
9.2	Artificial Stream Channel (drainage or irrigation ditch or stream diversion channel) - shallow open water dominated by submergent aquatics
9.3	Natural Perennial or Semi-perennial Stream Channel - dominated by emergent (usually Deep Marsh) wetland vegetation

Table 2.4 (cont'd)

Code No.	Wetland or Depression Type
9.5	Natural Perennial or Semi-perennial Stream Channel - shallow open water dominated by submergent aquatics
9.6	Natural Perennial Stream Channel - unvegetated water-washed bed
9.8	Artificial Lake - large deep impoundment
9.9	Natural Lake

Table 2.5. Densities (numbers per square kilometer) of various types of wetlands summarized by vegetation zone[a].

Landform	Stratum	Number of Air/Grounds per Stratum	Area Surveyed (in km^2)	Low Prairie to Wet Meadow[b], Excluding Type 0.3	Unclassified (0.3)	Shallow Marshes and Better[c]	Total Basins[d]	Potential Density[e] of Shallow Marshes and Better	Estimated Density of Total Basins under Pristine Conditions
Parklands	26	9	104.2	9.7	11.7	24.9	46.4	28.3	38.7
	30	4	29.5	11.8	14.2	8.7	34.7	11.2	32.3
	31	4	33.0	5.5	21.1	14.7	41.3	21.1	36.6
	34	5	44.0	12.7	29.9	25.9	68.4	34.6	64.7
	38	2	22.3	26.2	2.6	6.2	34.9	7.0	29.5
	40	4	25.2	8.8	18.3	26.9	54.0	32.3	47.9
Total/Average		28	258.2	11.2	16.2	20.5	48.3	25.1	42.2
Mixed prairie	27	3	35.0	10.9	6.8	15.0	32.7	17.0	27.5
	28	3	35.0	3.8	9.7	8.0	21.5	10.8	18.0
	32	10	76.4	5.2	11.7	13.8	30.6	17.6	26.5
	35	4	40.8	6.6	16.8	17.1	40.5	22.0	36.1
	39	5	47.3	7.3	13.0	17.0	37.5	20.8	33.4
Total/Average		25	234.5	6.5	11.8	14.3	32.7	17.9	28.4
Shortgrass prairie	29	4	46.7	4.2	4.6	5.9	14.7	7.3	7.8
	33	3	27.8	2.2	6.3	4.2	12.7	6.2	11.1
Total/Average		7	74.5	3.4	5.2	5.3	14.1	6.9	8.9
Prairie Total/Average		60	567.2	8.3	12.9	16.0	37.4	19.7	32.2

[a] Modified from Miller 1982, 1983, 1984

[b] Types 0.1 to 1.9

[c] Types 2.1 and better

[d] Excluding totally destroyed wetlands (Type 0.0)

[e] Based on the number of type 0.3 wetlands which, it is estimated, would be Type 2 or better under natural conditions

Seasonal and annual variability in pothole density is high. The average number of potholes at Lousana and Redvers decreased by 17.8 and 31.3%, respectively, between May 1 and July 1; over 80% of the potholes lost were less than 0.5 acres (.2 ha) in size (Table 2.2). These seasonal reductions vary over the pothole region: although densities in the parklands are twice those in grasslands in May, in July the parklands have three times as many potholes due to much greater rates of evaporation and the subsequent drying of potholes in the grasslands (Smith *et al*. 1964).

The cyclical occurrence of drought in the prairies also contributes to inter-year variability in pothole density: in an average year one-third of wetlands dry up, while in a dry year as many as two-thirds may disappear (Smith *et al*. 1964). At Redvers, Saskatchewan, average pothole densities during drought years were approximately one-half the densities recorded during non-drought years (Table 2.6). A reduction in pothole numbers one year due to drought conditions cannot be simply correlated with low levels of precipitation in the same year. The water dynamics of potholes is complex and the interpretation of current water levels requires an examination of levels of precipitation over past years, spring runoff events and soil moisture conditions.

2.1.5 Water regime

The high variability in pothole densities is reflected in wetland classification schemes where permanency of the water body is a major criterion (Martin *et al*. 1953, Stewart & Kantrud 1971, Millar 1976). Five major types of potholes can be distinguished on this basis:

a) *ephemeral potholes* where water disappears in a few days to a few weeks and the vegetation in the basin is characterized by wet meadow species;

b) *temporary potholes* which hold up to 18 inches of water for 4-12 weeks, and where the vegetation is composed of emergents such as rushes, sedges and cattails;

c) *seasonal potholes* which may dry out during droughts but generally persist through summer and where the vegetation consists mainly of emergents;

d) *semi-permanent potholes* which contain 2 to 5 feet of water and do not dry out for several years, and where both emergents and submergents are found in abundance;

e) *permanent wetlands* where water can be as deep as 10 feet and which have the same vegetation as semi-permanent potholes.

Table 2.6. Average density of potholes (number per km^2) at Redvers, Saskatchewan, during drought and non-drought years[a].

Time Span	Time of Year			
	May 1	June 1	July 1	August 1
mean pre-drought years 1952-1958	26.2	23.2	20.8	15.4
mean drought years 1959-1963	15.0	13.5	9.3	6.6
mean post-drought years 1964-1965	36.7	24.3	18.5	12.0
all years 1952-1965	23.5	19.7	16.2	12.0

[a] Modified from Smith and Stoudt 1968

The permanency of a wetland is determined by its water regime. Because of the lack of drainage in the prairie pothole region, inflow of water consists mostly of snowmelt, rainfall and groundwater, while water loss is due mainly to evaporation and, to a lesser degree, groundwater seepage. The amount of water reaching a pothole will be a function of precipitation, soil moisture, temperature, wind and the size and nature of the watershed. The time, intensity and duration of snowmelt along with the weather in the spring, level of moisture at freeze-up and depth of frost penetration in the fall, all determine the proportion of snowmelt which will reach the slough as surface runoff (Millar 1969).

Soil condition is extremely important in controlling inflow. For instance, frozen wet soils in the spring allow more snowmelt to run off into potholes than do dry soils. However, soil moisture in spring is a function of the previous fall's precipitation. The stage of a slough is frequently a measure of the winter regime of several seasons past (Eisenlohr 1969).

In central Saskatchewan, where average annual precipitation equals one-half the average annual evaporation over bodies of water, evaporation demand exerts a primary control on the permanence of a slough (Eisenlohr 1969). Under such conditions, the size of the drainage basin is critical for providing sufficient inflow to prevent total water loss from the pothole during the summer. Emergents decrease water loss by sheltering the water surface from wind and sunlight but also increase it through evapotranspiration. The net effect of vegetation was to increase water loss by 10% in one pothole when compared to a pond free of vegetation (Eisenlohr 1966).

Investigating factors which influence the depth and permanence of potholes, Millar (1969, 1971) found that the rate of water loss is directly related to the length of shoreline per unit water area. As smaller sloughs have more shoreline per unit area, they tend to lose water at a greater rate. Three factors account for this shoreline-related water loss: transpiration by vegetation in the margin of the slough, evaporation along the shoreline where water is heated to higher temperatures, and losses to groundwater. Sixty to eighty percent of shoreline-related water loss was the result of evaporation and transpiration processes with the balance due to groundwater loss (Millar 1971). In sloughs smaller than 0.1 acres (0.04 ha), shoreline-related losses accounted for 60% or more of total water loss while, in larger sloughs, it accounted for no more than 35%. Other losses were through water evaporation from the pond surface.

Groundwater flow systems have an appreciable effect on pothole hydrology and salinity (Sloan 1970, 1972). The surface of the water table is shallow, follows the general topography and is continuous with the surface of the prairie potholes. The direction of groundwater flow is a function of the height of the water table relative to that of the slough water surface: groundwater inflow, throughflow and outflow will occur when the water table adjacent to the pothole is respectively higher, intermediate and lower (Figure 2.5). Due to the lack of a drainage system linking pothole complexes, seepage outflow is the major route for the removal of dissolved salts. Net outflow results in less saline temporary sloughs while net inflow gives rise to more permanent water bodies with higher concentrations of salts. Meyboom (1966) demonstrated that, in south-central Saskatchewan, surface water recharges groundwater in spring and early summer, while in the latter part of the season groundwater replenishes small temporary sloughs; the net effect over one year was a net contribution to the regional groundwater.

2.2 Water and Soil Chemistry of Potholes

The physical characteristics of prairie potholes have a direct bearing on the chemistry of the water and soil of these systems. In addition, factors such as the water regime, the weather and the composition of surface deposits influence the chemical profile of the potholes and their behaviour under changing conditions. It is, therefore, important to determine the chemical profile and its variability in space and time as a number of chemical variables will influence the fate, persistence and toxicity of contaminants entering potholes.

2.2.1 Water chemistry

The range of values recorded for ten chemical variables in prairie potholes is presented in Table 2.7. There is a large amount of variability, within and between studies, in the concentrations of the variables except for pH. The potholes are generally slightly to moderately alkaline (pH 7.7 to 9.7) except for temporary sloughs which tend to be slightly acidic (Driver 1977, Driver & Peden 1977). The greater acidity of temporary sloughs may be related to the presence of humic compounds associated with the dense vegetation and decaying plant matter (Dvorak 1970a) which dominate these small and very shallow temporary water bodies. There is little variation from the mean pH recorded in each of the studies; Driver and Peden (1977) noted that the pH varied less than 0.5 units diurnally.

Conductivity or specific conductance is the reciprocal of the specific resistance of a solution measured between two electrodes 1 cm^2 in area and 1 cm apart, at a standard reference temperature of 25°C (Wetzel 1975). Conductivity is measure of the content of ionized salts or salinity: Barica (1975a) showed a very high degree of correlation between conductivity and total ion concentration (correlation coefficient = 0.99) in saline eutrophic lakes of Manitoba. The high variability in salinity of potholes between sites in the prairie pothole region is evident from Table 2.7. Driver and Peden (1977) found that conductivity changed from spring thaw to late summer in 80 small prairie ponds: salinity increased 67% in temporary ponds, 63% in semi-permanent ponds and 20% in permanent ponds. These changes were related to evaporation, transpiration, seepage and precipitation.

Figure 2.5. Relation of the direction of groundwater flow (arrows) and water-table configuration (from Sloan 1972).

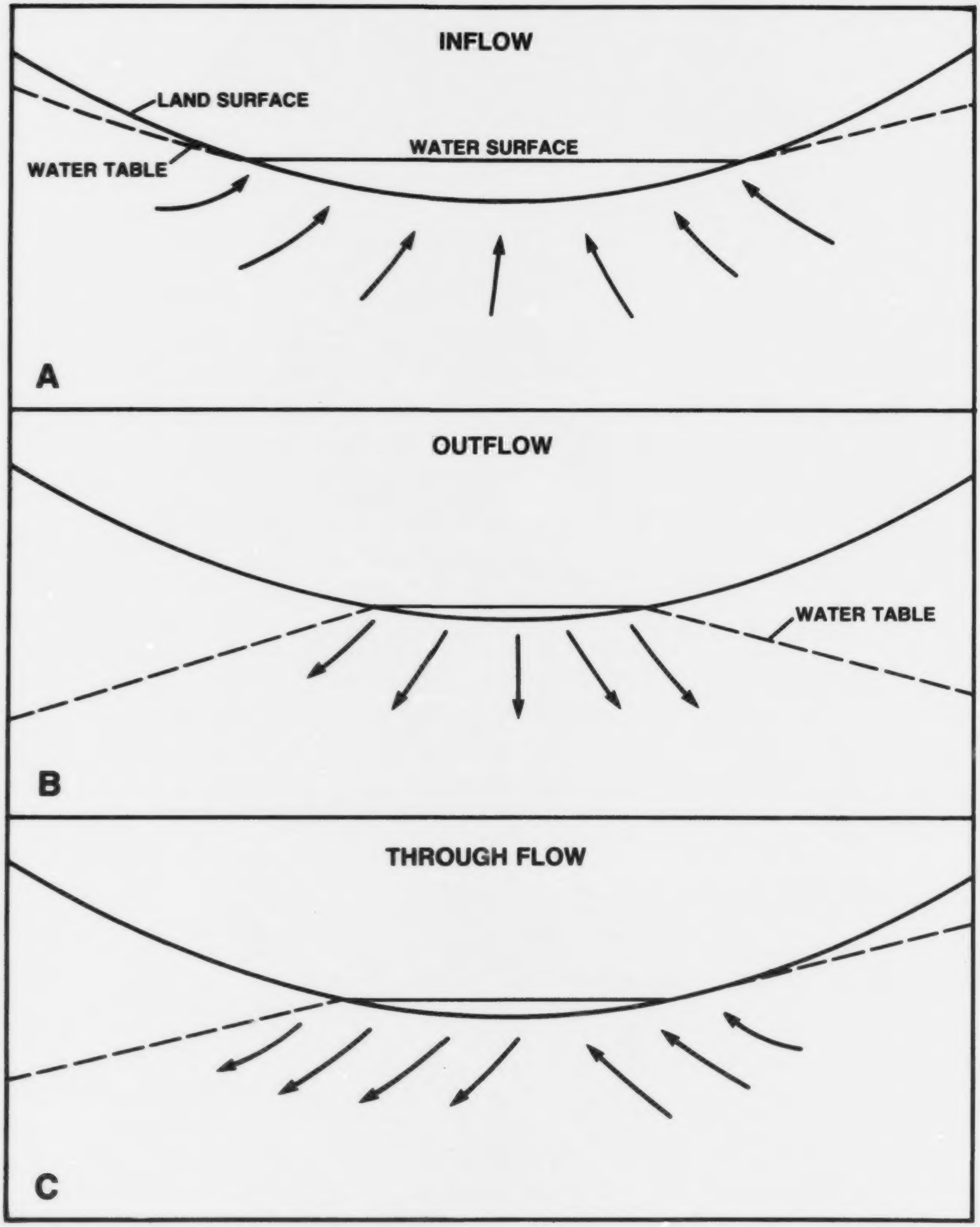

INFLOW
LAND SURFACE
WATER SURFACE
WATER TABLE
A
OUTFLOW
WATER TABLE
B
THROUGH FLOW
C

Table 2.7. Pothole water chemistry from various sites in the prairie pothole region.

		Concentration in ppm									
Location	pH	Conductivity (μmho/km)	Total Alkalinity ($CaCO_3$)	HCO_3	Ca	Mg	Na	K	Cl	SO_4	Source
4 sloughs near Wainwright, Alberta[a]	9.4-9.7 9.0 8.7 8.9-9.0		400-410 810-820 570 590		97-100 35-36 71-72 37-38	420 300-320 230-240 260-270	660-670 500-510 380-390 520-530	70 40 60 46			Shaw, C.W.S., unpublished data
12 seasonal wetlands in drift plain, North Dakota[b]	8.0±0.3	528±385	223±71		41±19	32±29	25±26	24±5	20±8	150±162	Swanson *et al*. 1974
12 seasonal wetlands in stagnation moraine, North Dakota[b]	8.1±0.2	473±209	134±27		50±29	18±13	7±5	14±3	13±3	105±83	"
12 lakes in outwash moraine, North Dakota[b]	9.0±0.3	32583±12826	1743±1020		54±43	1792±1335	7518±3572	709±338	3176±2044	17180±12203	"
12 lakes in stagnation moraine, North Dakota[b]	8.3±0.3	1234±492	390±85		55±15	94±37	78±65	38±13	28±15	276±233	"
10 temporary ponds near Floral, Saskatchewan[b]	6.8±0.3	551±731		155±117	57±82	23±40	21±39	27±20	10±12	167±358	Driver 1977

Table 2.7 (cont'd)

		Concentration in ppm									
Location	pH	Conductivity (μmho/km)	Total Alkalinity ($CaCO_3$)	HCO_3	Ca	Mg	Na	K	Cl	SO_4	Source
5 semi-permanent ponds near Floral, Saskatchewan[b]	7.7±0.4	1041±735		237±51	87±35	57±50	72±84	24±6	24±15	387±415	Driver 1977
1 permanent pond near Floral, Saskatchewan[b]	8.3	394		159	46	11	7	20	9	56	"
7 potholes, South Dakota[b]		333±196	130±37				15.7±24.4	11.6±3.6	2.4±1.1	52±59	Drewien and Springer 1969
79 saline winterhill pothole lakes in southwestern Manitoba[a]	7.4-9.6	305-7837			28-336	9-1236	2-980	11-248	1-191	12-6571	Barica 1975a
39 wetlands near Minnedosa, Manitoba[b]	8.5±0.4	2209±1332	280±82		111±48		117±109	38±13		1128±1015	Adams, C.W.S., unpublished data
53 wetlands near St-Denis, Saskatchewan[b]	8.5±0.4	1425±1996	301±116		96±65		76±209	35±24		640±1435	

[a] Range of values recorded
[b] Mean and standard deviation

Alkalinity, measured as concentrations of $CaCO_3$ or HCO_3, is also highly variable (Table 2.7). Alkalinity does not seem to increase with increasing conductivity which may be due to the poor correlations between conductivity and the ions Ca, HCO_3 and CO_3 as reported by Barica (1975a) for saline lakes of Manitoba. In contrast, the other anions SO_4 and Cl and the cations Mg, Na and K correlate well with conductivity. This would explain the change in ionic dominance patterns as a function of conductivity in 80 small prairie wetlands (Driver & Peden 1977). Table 2.8 indicates that bicarbonate is the dominant anion in potholes with conductivities less than 1000 µmho/cm while sulphate dominates above that. Similarly, in potholes with greater conductivity, Ca and K are replaced by Mg as the dominant cations. Seasonally, ionic dominance patterns did not change in spite of changes in salinity due to evaporation.

Nutrient concentrations (various forms of N and P) vary considerably within a pothole during a season, due mostly to biological processes. Driver and Peden (1977) found nitrate concentrations between 10 and 800 µg/l while total dissolved phosphorus did not exceed 300 µg/l in small potholes in Manitoba and Saskatchewan; in these same ponds total organic carbon ranged from 8 to 108 µg/l.

The high variability in pothole chemistry within and between regions can be attributed to two major factors: the geology and the influence of groundwater. Driver and Peden (1977) noted that ponds high in the watershed were usually fresh (<600 µmho/cm) and those low in the watershed usually saline (2000-23,000 µmho/cm). This reflects the involvement of groundwater in the water regime as discussed earlier. The same authors also found that chemical profiles of potholes within study areas were not sufficiently unique to permit differentiation of geographic areas." They pointed out that this was partly due to the heterogeneous nature of glacial deposits. The heterogeneity is illustrated in conductivity measurements presented for four sites in North Dakota (Table 2.7): lakes in glacial outwash were almost 100 times more saline than lakes or seasonal wetlands in either stagnation moraine, ground moraine or drift plain.

2.2.2 Soil chemistry

Pothole bottoms have been called both soils and sediments due to the alternately terrestrial and aquatic nature of these wetlands. In the smaller temporary potholes and in the shallows of larger sloughs, the bottom is often dry for a good part of the year and consequently resembles terrestrial soils. In deeper sloughs, the bottom tends to be composed of finer sediments such as those of lakes.

Table 2.8. Ionic dominance patterns as a function of conductivity in 80 small prairie wetlands[a].

Ions	Conductivity (μmho/cm)
Anions:	
$HCO_3 > Cl > SO_4$	44-50
$HCO_3 > SO_4 > Cl$	500-1,100
$SO_4 > HCO_3 > Cl$	600-1,200
$SO_4 > Cl > HCO_3$	>12,000
Cations:	
Ca-K	<300
Ca-Mg	300-1,100
Mg (except for 3 ponds where Na dominates)	>1,100

[a] Driver and Peden 1977

Of 43 sediment samples from open-water areas of semi-permanent potholes in Manitoba, 44 and 28% were classified as loam and clay, respectively (G. Adams, CWS, unpublished data). Overall organic matter content averaged 7.4%; loam soils had the highest organic matter content with an average of 10.8% while clay soils averaged 3.9% and the one sandy bottom had 0.6%. Concentrations of NO_3-N, available P and K averaged 3.7 ± 4.3 (1 S.D.), 11.2 ± 11.8 and 542.5 ± 164.9 ppm, respectively. One would expect much greater amounts of organic matter, due to slower rates of decomposition, in soil samples taken from shallower and more densly vegetated areas of the potholes (Polunin 1984). For example, organic matter content exceeded 75% in a marsh in Wisconsin (Lee et al. 1975).

2.3 The Vegetation in and Around Potholes

The prairie pothole region of Canada includes both grassland and parkland (see Figure 2.3). The latter is a transition zone between the aspen forest of the north and the grasslands to the south. It is characterized by clumps of trees surrounded by various shrubs and grassland species; as one moves from south to north, the abundance of trees increases to a point where the forest becomes continuous.

In areas of relatively little cultivation or grazing within the parklands, the woody growth consists mostly of groves of quaking aspen, Populus tremuloides, balsam poplar, Populus balsamifera, and, in southeastern Saskatchewan on drier sites, burr oak, Quercus macrocarpa. These groves are usually surrounded by various species of shrubs such as buckrush, Symphoricarpos occidentalis, snowberry, S. albus, choke-cherry, Prunus virginiana, saskatoon, Amelanchier alnifolia, or cranberries, Viburnum sp.

While the occurrence of grassland species is strongly influenced by the degree of present and past disturbances (e.g. cultivation, grazing or burning), the species most commonly noted are bluegrasses, Poa sp., foxtail, Hordeum jubatum, wheat grasses, Agropyron sp., the sedge Carex eleocharis and oat grasses, Danthonia spp. (Smith & Stoudt 1968).

The grassland species cited above are characteristic of the uplands which are defined as the areas beyond the margin of the pothole or the high water mark following the spring snow melt. A more extensive list of the important plant species of the uplands is found in Table 2.9. This vegetation zone is very important within the agricultural landscape for providing adequate nesting cover for breeding ducks as will be discussed in Section 2.5.

A description of the mammalian and avian fauna of the parklands can be found in Smith (1971) and Stoudt (1971) and will not be examined in this section. It should be pointed out, however, that the species composition reflects the heterogeneous make-up of the area: both grassland and forest species are found together. Of interest are those birds, other

Table 2.9. Species of vegetation common in uplands and wetlands of the Canadian prairies[a].

COMMON NAME	SCIENTIFIC NAME
Upland and margin vegetation:	
quaking aspen	*Populus tremuloides*
balsam poplar	*Populus balsamifera*
burr oak	*Quercus macrocarpa*
buckbrush	*Symphoricarpos occidentalis*
snowberry	*S. albus*
willows	*Salix* spp.
choke-cherry	*Prunus virginiana*
pincherry	*P. pennsylvanica*
Saskatoon	*Amelanchier alnifolia*
cranberries	*Viburnum* sp.
woodrose	*Rosa woodsii*
hazelnut	*Corylus cornuta*
wild currents	*Ribes* spp.
meadowsweet	*Spiraea alba*
anemone	*Anemone canadensis*
violets	*Viola* spp.
northern bedstraw	*Galium boreale*
aster	*Aster* spp.
bunchberry	*Cornus canadensis*
thistle	*Cirsium* spp.
sow thistle	*Sonchus* spp.
foxtail	*Hordeum jubatum*
quack grass	*Agropyron repens*
other wheatgrasses	*Agropyron* spp.
sweet clover	*Melilotus* spp.
bluegrasses	*Poa* spp.
needlegrass	*Stipa* spp.
grama grass	*Bouteloua gracilis*
brome	*Bromus inermis*
Emergent marsh vegetation:	
rush	*Juncus balticus*
sedges	*Carex* spp.
whitetop	*Scolochloa festucacea*
spike rush	*Eleocharis palustris*
cattail	*Typha latifolia*
softstem bulrush	*Scirpus validus*

Table 2.9 (cont'd)

COMMON NAME	SCIENTIFIC NAME
hardstem bulrush	*S. acutus*
alkali bulrush	*S. paludosus*
manna grass	*Glyceria* spp.
water smartweeds	*Polygonum* spp.
water parsnip	*Sium suave*
water plantain	*Alisma triviale*
arrowhead	*Sagittaria* spp.
golden dock	*Rumex maritimus*
mare's tail	*Hippuris vulgaris*
common reed	*Phragmites communis*
Floating vegetation:	
common duckweed	*Lemna minor*
star duckweed	*Lemna trisulea*
Submergent marsh vegetation:	
pondweed	*Potamogeton* spp.
water milfoil	*Myriophyllum exalbescens*
hornwort	*Ceratophyllum demersum*
bladderwort	*Utricularia vulgaris*
water-crowfoot	*Ranunculus* spp.
widgeon grass	*Ruppia* spp.
water starwort	*Callitriche palustris*

[a] List made from species cited in Smith and Stoudt 1968, Millar 1976 and Driver 1977

than waterfowl, which are associated with wetlands. In a study of 22 common wetland-dwelling bird species, Kantrud and Stewart (1984) reported that temporary, and especially semi-permanent potholes, supported the greatest diversity of birds and most of the avian population (Table 2.10). This was attributed to the greater vegetative diversity of these two pothole types (see Section 2.2.2).

2.3.1 Vegetation zones

Due to the dynamic nature of water levels in potholes, changes in the composition of the vegetation are better indicators of the boundary between prairie and wetland than is the waterline. This is reflected in Millar's (1976) definition of a wetland as "that portion of a basin which is normally covered with shallow water for at least a portion of each year and, in an undisturbed state, supports wetland vegetation ...."

The physical appearance of a pothole is strongly determined by the type of plant species present and their distribution within the pothole. The physical form of the plants strongly affects the structure of the invertebrate community and the dynamics of the whole ecosystem. Plant form is indicative of water depth and duration of flooding: emergents have rigid stems and leaves to support the bulk of the biomass which is aerial, while submergents tend to have more flaccid structures. Therefore, plant form is used here to distinguish between three different types of vegetation: the emergents, the submergent and floating plants, and the algae. The first two categories of plants have been used in combination with other characteristics of potholes to classify wetlands (Stewart & Kantrud 1971, Millar 1976). These classification schemes have divided wetlands into two broad categories of vegetation zones: the emergent vegetation zones and the open-water zones where submergents dominate.

2.3.1.1 Emergents species

A list of the dominant emergent species in prairie wetlands can be found in Table 2.9. Three emergent vegetation zones can be delineated on the basis of species composition: the wet meadow, shallow marsh and deep marsh zones. The wet meadow zone is usually found in the central portion of shallow basins or at the periphery of deeper wetlands where flooding lasts 3 to 4 weeks after spring runoff or several days after heavy rainfall in summer. The vegetation is characterized by fine-textured grasses and sedges, forbs and occasionally willow shrubs. This zone has the highest plant species diversity: Millar (1976) reported 75 species. The shallow marsh zone, on the other hand, usually goes dry by late summer. Its vegetation is characterized by grasses, sedges and forbs of greater height than those in the previous zone and species richness is much lower: 33 species were noted by Millar (op. cit.) The deepest of the three emergent

Table 2.10. Distribution (%) of breeding birds among classes of wetlands[a].

Species	Pairs in Sample N	Ephemeral	Temporary	Semipermanent	Permanent	Alkali	Fen	Undifferentiated Tillage
Horned Grebe	11		64	36				
Eared Grebe	35		54	34	11			
Western Grebe	20			5	95			
Pied-billed Grebe	111		23	68	7		1	
Black-crowned Night-heron	36		6	75	19			
American Bittern	54	2	30	69				
Northern Harrier	13		23	77				
Virginia Rail	14		14	86				
Sora	154	1	45	54				
American Coot	1,648	tr[c]	24	72	4			
Killdeer	51	4	53	39	4			
Willet	118	2	56	39	3	1		
Marbled Godwit	65	3	57	37		3		
American Avocet	45		38	47		9		7
Wilson's Phalarope	257	4	60	30		4	tr	3
Black Tern	410	tr	25	72	2		tr	
Marsh Wren	193		9	91				
Common Yellowthroat	60		48	52				
Yellow-headed Blackbird	1,104	tr	6	94				
Red-winged Blackbird	1,245	2	61	35	1	tr	tr	
Savannah Sparrow	167	11	49	38		1	1	
Total	5,811	1	33	63	2	tr	tr	tr

[a] Modified from Kantrud and Stewart 1984
[b] See Table 2.4 and Section 2.1.5 for definition of wetland classes
[c] tr = < 0.5%

vegetation zones, the deep marsh zone, usually holds water until fall if not through the winter. Floristically, this is the poorest of the three zones with five species recorded by Millar (op. cit): Typha latifolia, three species of Scirpus and Phragmites communis. These coarse, grass-like plants are much taller than corresponding species in the shallow marsh zone.

The three zones described above reflect increasing depth and duration of flooding. These zones are usually arranged as concentric rings with the wet meadow zone at the periphery of the wetland and the deep marsh zone closest to the central portion of the basin (Figure 2.6a). As the basin gets deeper, the emergent vegetation is replaced by open water where submergent and floating species are favored. Often, open water areas are found interspersed within the marsh zones.

2.3.1.2 Submergent and floating species

Submergents are rooted plants which have most of their leaves and stems under water but form flowers and fruits at the surface. A list of common species found in prairie potholes is presented in Table 2.9. Their leaves are fine, complex and compound in order to increase light gathering efficiency under conditions where light is limiting. The floating plants are not rooted and are found at the water surface. In the Canadian prairies, these are small and consist mostly of the duckweed Lemna.

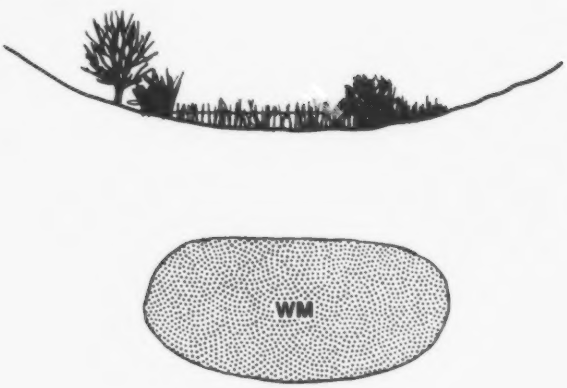
Open water zones where submergent and floating species are dominant are usually permanently flooded, but are at least flooded for several years at a time. They are shallow enough to allow the growth of rooted aquatic plants. Certain open water zones are transitional in nature and result from the destruction of stands of emergents by high water levels. Given enough time, submergents will dominate these zones if continuous flooding persists. The type of submergent species found in this zone is indicative of the duration of flooding: certain aquatic species, such as the pondweed Potamogeton pusillus, do not appear until after two or more years of continuous flooding (Millar 1976). Open alkali zones are free of emergents due to high concentrations of alkali salts which are tolerated by a single species of macrophyte, the submergent Ruppia maritima.

2.3.1.3 Algae

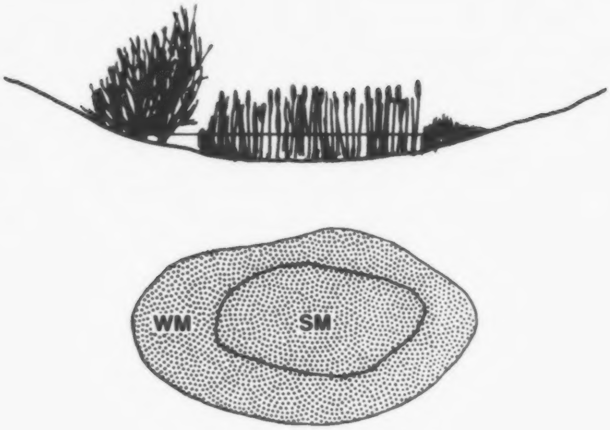
The algae of wetlands can be divided into two broad categories: the unattached phytoplankton and the periphyton. Except for their contribution to total primary production and as food to invertebrates, few studies have attempted to evaluate the extent of the influence of either group on ecosystem dynamics.

Figure 2.6a. Vegetation patterns in stable wetlands shown in vertical and profile diagrams. Each wetland type is given the name of the vegetation zone which occupies its center except for the open water march which has a central shallow open water zone (taken from Millar 1976).

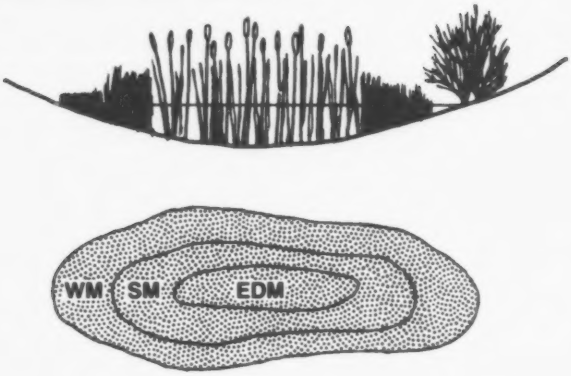
WET MEADOW



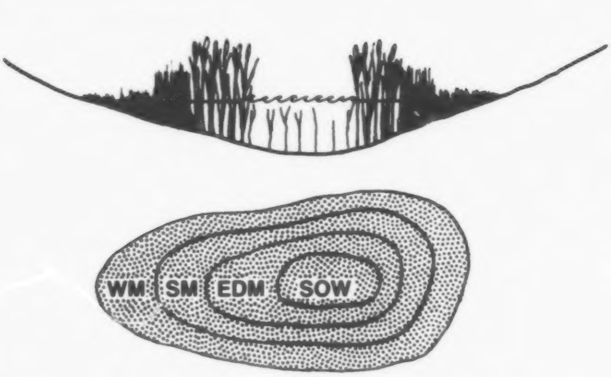
SHALLOW MARSH



EMERGENT DEEP MARSH



OPEN WATER MARSH


WM
SM
EDM
SOW

Very large blooms of the blue-green algae are known to occur in mid-summer in the open water and in small openings in the vegetation of wetlands. The blue-green alga Aphanizomenon flos-aquae was responsible for most of the late July blooms in 50 small prairie lakes of the Erickson-Elphinstone district of southwestern Manitoba (Barica 1975a, Kling 1975). These blooms were responsible for the oxygen depletion encountered after their collapse. While the lakes studied were larger and deeper than potholes, they were self-contained, relatively shallow and highly eutrophic systems and therefore are comparable to smaller sloughs. Large blooms followed by massive die-offs and anaerobic conditions probably play an important role in determining the availability of nutrients and the amount of light penetrating the waters and thereby influence macrophyte production.

A study of epiphytic algal growth on cellulose acetate substrates in a marsh pond showed that the algae belonged to only three taxonomic groups: Chlorophyceae, Bacillariophyceae and Cyanophyceae (Hooper-Reid & Robinson 1978a). The greens and diatoms were the most important contributors to total cell volume and surface area. Biomass generally increased steadily throughout the growing season, reaching peak values of 10-14 g dry weight per m^2 of substrate in early fall. The study also showed how growth differed with the site of the substrate in the marsh: epiphytic growth within stands of the submergent Potamogeton pectinatus showed a large increase in mid-July followed by a decrease to a value which then remained relatively constant until the fall peak. In stands of the emergent Scirpus acutus, biomass increased slowly until a peak in October. The mid-July peak in the Potamogeton site was mostly the result of blue-green algal growth. A further study (Hooper-Reid & Robinson 1978b) indicated that nutrient availability limited algal growth and that differences in nutrient levels between stands of Scirpus and stands of Potamogeton were responsible for differences in biomass through the growing season. In an earlier study, Hooper and Robinson (1976) found that production was higher on emergents than submergents and concluded that light was a limiting factor to epiphyte growth.

2.3.2 Factors affecting plant distribution within potholes

As one moves from the wet meadow zone to the center of a wetland, the succession of distinct vegetation zones in the form of concentric rings (Figure 2.6a) parallels a gradient of increasing depth and duration of flooding. Therefore, the type of vegetation in the center of the basin, its density and its vitality, reflect the flooding history and are key factors used in the classification of wetlands (Millar 1976). Association analysis of plant species frequency and presence in 246 stands sampled in 18 study areas of Saskatchewan showed that among eight environmental variables - water depth, pH, dissolved solids, texture and organic content of soil, water regime, salinity and disturbance (mowing, grazing, cultivating) - the last three, in that order of importance, were the factors most influencing species distribution (Walker & Coupland 1970).

2.3.2.1 Water regime

The importance of water depth as a factor in wetland species distribution has been highlighted by work done on vegetation-environment relationships in Saskatchewan sloughs (Walker & Coupland 1968, 1970). The highly variable water regime of prairie sloughs (Section 2.1.5), coupled with the close relationship between water regime and plant species distribution, render these wetlands extremely variable with respect to vegetation structure on a seasonal and yearly basis. Millar (1973) followed changes in the structures of the common plant communities of 71 shallow marsh wetlands in Saskatchewan between 1962 and 1971, during which time water conditions improved from an initial drought situation. These wetlands typically go dry by mid-summer and consist mostly, at their deepest point, of grasses and grass-like plants. During the 10-year study period, 46% of the stands experienced changes in vegetation density; while 75% retained shallow marsh characteristics, one-quarter converted to open water. Greater than normal water depths induced partial die-offs but overwinter flooding was required to produce total eradication of shallow marsh emergents: none survived more than two years of continuous high water levels. However, deep marsh emergents did not appear in a majority of the stands because requirements for establishment are more complex than mere improvements in the moisture regime. Submergents developed in approximately 50% of the stands and the type of species which became established was a function of the duration of flooding. Open deep marshes rarely developed in basins less than one acre (.4 ha) in size because of higher rates of water loss in the smaller sites. In situations where an emergent deep marsh is subjected to extreme flooding, the shallower zones may die out first, thus creating an inversion of the normal pattern with a central deep emergent zone surrounded by open water (Millar 1976).

During periods of drought, vegetation zones die-off but reestablish themselves in deeper areas of the marsh while maintaining the same pattern of zonation. Deviations from the expected pattern can occur when open water areas go dry long enough for shallow marsh emergents to establish themselves inside the emergent deep marsh zone (Millar 1976).

As water levels in a pothole increase, there is both a decrease in light reaching the plants and an increase in pressure. A drop in luminosity will certainly have a negative effect on growth. Increased pressure results in growth reduction in Nuphar advenum (Laing 1941). A study conducted to examine growth and reproductive responses of the emergent Scirpus maritimus to water depth clearly demonstrated increasing energetic costs to the plant with higher water levels: shoots rising from tubers were taller and less numerous while underground and total biomass was significantly reduced (Lieffers & Shay 1981). In addition, the plants shifted from clonal growth to seed production, which indicates the presence of unfavorable conditions. Thus, the potential for die-offs through exhaustion of energy supplies exists at high water levels.

Experimental work by Laing (1941) and a study of massive emergent die-offs on the shores of Lake Erie (McDonald 1955) suggest that anaerobic conditions may be detrimental to water plants. Exceptionally high waters during the winters of 1945-1946 and 1951-1952 in Lake Erie marshes submerged dormant shoots of the cattail Typha, thereby limiting the availability of oxygen and causing rapid die-offs. Although the rhizomes can tolerate anaerobic conditions for a short period, an entire winter without oxygen was too long.

The outcome of massive losses of water plants due to higher than normal water levels may be an increase in plant species diversity. This occurred in Lake Erie marshes following the die-off of the dominant emergent, Typha (McDonald 1955). A succession of submergents and floating-leaf species took over areas formerly occupied by the cattail. When water levels fell again, a variety of wetland plants colonized these sites. The emergent vegetation was fragmented into much smaller stands as a result of the die-off, thereby increasing the amount of edge between the vegetation and open water. All these changes were beneficial in providing adequate foods and a more diverse habitat for waterfowl and other birds inhabiting the marsh.

Flooding also favors the propagation of certain species: willow seedlings require surface moisture until the root system penetrates to the water table (Millar 1969). Scirpus maritimus shifted from clonal propagation to seed production with increasing flooding (Lieffers and Shay 1981). Although these conditions may lead to the death of the plant, flowering and seed production allows the colonization of new growing sites and assures long-term survival.

Decreases in water level can also bring about changes in reproductive strategy. Certain submergents such as Utricularia spp. and Myriophyllum verticillatum flower more and depend less on vegetative growth in shallow water (Sculthorpe 1967). Exposure of mudflats because of drawdowns (Kadlec 1962) or drought (Millar 1969) resulted in the spread and increased abundance of emergent species. As with willow species, the moisture requirements during the germination and the seedling stages of these emergents are different from those of the mature plants. Many of the species require exposed soil for germination and early growth while subsequent survival is limited to basins which hold water. Further benefits of drawdowns are increased availability of nutrients and thus, increased production. Experimental reduction of water levels in a Michigan marsh resulted in increases of nitrogen due to aerobic nitrification, greater concentrations of phosphorus in the water and increases in potassium in the soil solution (Kadlec 1962). This mobilization of nutrients from the soil may have been the source of the luxuriant growth of water smartweed and bushy pondweed which followed the drawdown.

2.3.2.2 Salinity

While water depth was the environmental variable which most affected the distribution of the leading dominants in 246 stands of wetland vegetation, salinity, as reflected by the conductivity of the water, was the next most important factor (Walker & Coupland 1970). Within vegetation zones, species composition is correlated with average salinity and this variable is used in numerous wetland classification schemes (Metcalf 1931, Mason 1957, Stewart & Kantrud 1971, Millar 1976). Salt concentrations fluctuate widely on a seasonal and yearly basis in a manner corresponding to water depth changes. As most wetland species tolerate some degree of fluctuation in salinity, the composition of the vegetation is a better reflection of average salinity than are single measurements of conductivity.

As salinity increases, species richness decreases; only the widgeon grass Ruppia maritima exists in hypersaline conditions. This may be due to a difficulty of other plants in adapting to higher concentrations of dissolved solids. In addition, the amplitude of fluctuations in salinity is positively correlated with average salinity values (Stewart & Kantrud 1971). Thus, salinity and its fluctuations may influence the establishment of certain plants as, in highly saline conditions, a plant species must not only tolerate high salt concentrations but also large fluctuations of this variable. The requirements of a species with respect to salinity can be different in the germination, seedling and mature stages (Millar 1969).

2.3.2.3 Disturbance

Disturbance which does not lead to the permanent elimination of the slough is the third most important factor influencing species distribution in wetlands (Walker & Coupland 1970, Stewart & Kantrud 1974, Millar 1976). Although, in the past, fires and fluctuations in water level were natural events, most disturbances today are caused by man. These consist of mowing, grazing, burning and various forms of cultivating (e.g. plowing, tilling, seeding, etc.)

Mowing, while it does not affect vegetation zone patterns, causes changes in floral composition depending on the timing of the practise. Annuals are the most affected as early mowing prevents seed production.

The impact of grazing depends on its intensity. The more palatable wet meadow and shallow marsh species are the first to be eliminated. If grazing is intense enough, these zones may disappear and lead to an inversion of the normal vegetation pattern with the deep emergent zone surrounded by open water.

A comparison of wetland vegetation composition in agricultural and pristine areas of the aspen parkland in Manitoba by Dwyer (1970) showed how emergents were affected by mowing and grazing. Sedges were less common in potholes within agricultural areas while grasses and forbs were favored.

The submergent vegetation was strikingly more abundant in wetlands situated within farmland than outside. The two dominant aquatic submergents, Ceratophyllum and Myriophyllum, frequently encountered in potholes within agricultural areas were absent outside these areas. This difference may have been caused by the inability of these plants to root in the loose undecomposed organic matter found on the bottom of sloughs which are subjected to inputs of leaf material from the forest surrounding them.

The effect of fire depends on the tolerance of wetland plants to such events and on whether the fire is fast-burning or slow and smoldering.

Removal of the brush ring around a basin through cultivation ultimately reduces the amount of runoff into the pothole by decreasing snow deposition (Millar 1969). This seriously diminishes the permanency of small sloughs. Furthermore, the cultivation of potholes causes a break-up of the organic layer accumulated on the bottom which acts as a "seal", and may lead to increased seepage outflow and a further reduction in wetland surface area. Millar (op. cit.) found that small cultivated potholes lost water much faster than undisturbed potholes of similar size. If not sown, cultivated potholes return to the first stages of secondary succession. If the basin stays dry, coarse weedy forbs will colonize the slough; if the basin is flooded again, perennial marsh plants will reestablish themselves (Ibid).

2.3.2.4 Soil properties

Physical and chemical properties of soil may play an important role in determining plant distributions in potholes by affecting establishment, growth and competitive interactions. For example, the distribution of sago pondweed, Potamogeton pectinatus, was significantly affected by soil texture in Delta Marsh, Manitoba (Anderson 1978). Available soil K and P, along with texture, were important factors affecting growth of the plants. Also, through their influence on soil distribution and turbulence, the intensity and direction of wave action affected the distribution of plants.

2.3.2.5 Consumers

Vertebrate and invertebrate consumers may have significant impacts on wetland vegetation. McDonald (1955) reviewed some records of massive die-offs of emergents which were attributed to either insect borers, aphids or microorganisms which caused root rot. The use of emergent marsh vegetation by muskrats for building lodges and as food has led to the elimination of all marsh plants during population explosions of these animals (Weller 1981). Although muskrats are present in the prairie pothole region of Canada, their impact on these systems is not well documented.

2.3.3 Nutrients and production

Nutrients are essential to plant growth and understanding their dynamics is, therefore, important. Unfortunately, little work has been done on nutrient cycles within prairie potholes. Two important characteristics of prairie sloughs sets them apart from other bodies of water with respect to nutrient dynamics and precludes direct extrapolation from knowledge gained on cycling in larger water bodies such as lakes: the lack of regular inflow or outflow and their shallow depth (generally <1 m). The former implies that alloctonous inputs of nutrients will be limited primarily to snowmelt and runoff events, with concentrations reflecting the fertility of the watershed. The shallowness of the potholes enhances the importance of the sediments as a source of nutrients and as a site for nutrient recycling. Of the numerous substances important for plant growth, only nitrate and phosphate will be discussed here because they are often the most limiting.

2.3.3.1 Nutrient import and export

The sources of nutrient input into wetlands are: surface runoff, rainfall, groundwater, the atmosphere and biomass. Runoff inputs are occasional events following rainfall or snowmelt in spring. The amounts and concentrations of nutrients carried with surface runoff are a function of land use practices and the size of the catchment area. Table 2.11 shows that the nutrient concentrations in spring runoff samples were highly variable between basins in southwestern Manitoba (Barica 1975a). This variation is most probably the result of differences in land use practices; forested areas will release few nutrients while high levels of dissolved organic nitrogen (DON) are characteristic of agricultural areas.

Rainfall also contributes to the nutrients of wetlands; it was found to contain more nitrogen and phosphorus than certain runoff waters (Kadlec 1979). However, nutrient losses from wetlands can occur through overflow following heavy rainfall. The extent of the flushing of nutrients from potholes will depend on the rate of water loss and the time of year this occurs: rapid turnover of the water after ice-melt may lead to significant losses of nutrients released from the sediments during the winter.

Because of the complexity of the hydrogeology of the prairies, little can be said about the contribution of groundwater to the nutrient budget of potholes. Groundwater nutrient concentrations are highly variable (NH_3-N conc. from trace to 3000 µg/l; NO_3-N conc. 0-10000 µg/l) and do not seem to be related to lake chlorophyl a levels in shallow lakes of southwestern Manitoba (Barica 1975a).

The exchange of nutrients between the atmosphere and wetlands is most important for the nitrogen cycle. While nitrogen enters wetlands via diffusion from the atmosphere as molecular nitrogen, fixation is the result

Table 2.11. Nutrient content of some spring runoff samples from five lake basins in the Erickson-Elphinstone district of southwestern Manitoba[a].

Lake Basin	NH_3-N	NO_3-N	DON[b] µg/liter	SRP[b]	DOP[b]	DIC[b]	DOC[b] Mg/liter	Si	Notes
103	159	79	-	12	-	6	-	-	snowmelt
885	204	122	-	40	-	-	-	-	"
103	309	10	2640	920	22	15.1	12.3	-	runoff
103	590	589	430	435	31	16.3	-	2.6	"
154	580	607	1320	309	41	18.2	-	5.6	"
003	29	-	1800	210	87	52.0	25.1	6.4	creek
005	4460	3870	2900	4950	0	-	18.0	5.6	barnyard runoff

[a] From Barica 1975a

[b] Abbreviations:

DON = dissolved organic nitrogen
SRP = soluble reactive phosphorus
DOP = dissolved organic phosphorus
DIC = dissolved inorganic carbon
DOC = dissolved organic carbon

of algal and bacterial action. The combination of a large blue-green algal biomass, high bacterial activity and warm waters results in characteristically high rates of nitrogen fixation in wetlands. Values of 0.6 to 6.0 g/m^2/yr have been reported (Kadlec 1979) and suggest that this may be a major route of nitrogen entry into potholes. In a similar manner, losses to the atmosphere can be substantial. Denitrification, the conversion of nitrate (NO_3) to N_2, is a microbial process occurring under anaerobic conditions. Loss of nitrogen through this pathway is favored in shallow fluctuating waters with alternating anaerobic and aerobic conditions and large amounts of decaying organic matter, conditions characteristic of a marsh. Export of nitrogen from marshes through denitrification is estimated to be of the same order as that being fixed, if not in excess of it (see Kadlec 1979).

The movement of animals in and out of wetlands can also make significant contributions to annual nutrient budgets. For example, not only can waterfowl consume a large fraction of macrophyte production, but they also excrete important amounts of nitrogen and phosphorus.

2.3.3.2 Compartmentalization of nutrients

Although an annual nutrient budget for potholes cannot be drawn up because of the lack of specific information, the compartmentalization of these nutrients within a slough can be inferred from work done on other wetlands. The rough approximations gathered in Table 2.12 indicate that there is at least one order of magnitude difference in the amounts of N and P between the three main compartments: sediment, biomass and water. The bulk of the nutrients are found in the sediments and, to a lesser degree, in plants. Water seems to hold little of the nutrients found in a wetland, probably due to either rapid uptake by algae and some plants or incorporation into the sediments. The concentration of N and P per unit area associated with plants is a function of the amount of plant biomass per unit area. This is why stands of emergent vegetation contain more nutrients than do stands of submergent vegetation in spite of greater concentrations in the tissues of the latter (Kadlec 1979). The nitrogen and phosphorus in sediments exists in two phases: dissolved in interstitial water or bound to particles. The particles exist in different forms which differ in their ability to release the nutrient ions. The availability of the nutrients in the sediments to plants depends strongly on the relative dominance of these various forms of binding and how they are affected by changes in the redox potential associated with fluctuating oxygen concentrations.

2.3.3.3 Cycling processes

Most of the processes involved in the cycling of nutrients in wetlands are reviewed by Kadlec (1979) and will not be discussed here in detail.

Table 2.12. Nitrogen and phosphorus concentrations in three wetland compartments.

Compartment		Concentration (g/m^2)		Source
		Total N	Total P	
Sediment		500-1500	24-800	Kadlec 1979
Biomass	Emergent	20-40	2-4	"
	Submergent	10-20	1-2	"
Water[a]		1-4	0.1-0.7	Kadlec 1979, Driver and Peden 1977, Barica 1975a

[a] Values were calculated using surface water nutrient concentrations and assuming a wetland depth of 1 metre

The physical-chemical processes consist of dissolution, precipitation, oxidation-reduction, adsorption, ion exchange, leaching and formation of complexes as influenced by pH, temperature and oxygen.

The microbiological processes consist of bacterial decomposition and oxidation-reduction. Bacterial activity is of major importance in recycling nutrients from the sediments to the water. This activity is affected by pH, carbon energy source and oxygen supply. The latter strongly influences the direction which the recycling of nitrogen will take. When oxygen is available, ammonia, an end product of decomposition, is oxidized to nitrate, while, under anaerobic conditions, nitrate is reduced to molecular nitrogen and NH_4 accumulates. NH_4 and NO_3 are available to plants and algae; N_2, however, is lost to the atmosphere.

The macrobiological processes are the result of animal and plant activity. Animals take up and release nutrients via ingestion and excretion. The rate of recycling of P per gram body weight is negatively correlated with body size. It has been suggested that zooplankton can release amounts of N and P equal to their total body content of these nutrients in 20 to 200 hours (see Allen & Kramer 1972). Lehman (1980) clearly demonstrated that zooplankton are the major source of N and P to the epilimnion of lakes during the stratified period. These studies indicate that invertebrates, which are abundant in prairie potholes, could be a major source of recycled nutrients within the sloughs.

Like animals, plants cycle nutrients through uptake (mostly from sediments) and excretion. Boyd (1970) measured uptake rates as high as 19.3 mg P/m^2/day and 162.7 mg N/m^2/day by the emergent Typha latifolia. However, plants take up nutrients in excess of their needs and this is reflected in the release of nutrients to the water: rates of P release ranging from 2.0 to 51.5 mg/m^2/day have been recorded (Kadlec 1979). It seems that, in light of these high rates of uptake and excretion, macrophytes act as nutrient pumps, extracting N and P from sediments and releasing them in the water. This pumping action of plants should lead to much higher nutrient concentrations in the water than those observed. The excess P is probably rapidly resedimented through chemical precipitation and algal uptake while N is possibly lost to the atmosphere after denitrification.

The evidence presented in the studies cited above indicates that marshes tend to accumulate nutrients. However, the nitrogen and phosphorus dissolved in water and available in the sediments is inadequate to provide for measured rates of plant uptake. Therefore, runoff inputs and processes involved in the release of nutrients from the sediments dominate these ecosystems. This is reflected in the correlation between wetland productivity and watershed fertility (Kadlec 1979).

The importance of autochthonous sources of nutrients in wetland ecosystem dynamics is highlighted in a study of nutrient cycling in shallow, self-contained, winterkill lakes (Barica 1974, 1975b). The term winterkill reflects the anoxic conditions under the winter ice cover. Under these conditions, there is an accumulation of inorganic N and P in the water as a result of anoxic bacterial decomposition of organic matter. As the ice cover melts and production resumes, high levels of nutrients support an excessive growth of algae which collapses when available nutrients in the water are exhausted. The die-off of algal blooms leads to increased bacterial decompositon and subsequent oxygen depletion. These two processes release nutrients back into the system thus triggering another algal bloom. This cycle repeats itself, in various degrees, throughout the growing season. Of major importance in these cycles is the depletion of oxygen which favors the bacterial mineralization of dissolved organic N and P to NH_3 and PO_4 which are the forms used by algae.

2.3.3.4 Production

Wetlands are among the most productive ecosystems (Table 2.14). The higher part of the range of production values for marshes is the same as that for cultivated land while the average production estimate of 2000 $g/m^2/yr$ is tied for first place with tropical rain forests. Two of the more important factors contributing to high marsh productivity are the availability of water throughout the growing season and the high nutrient levels in the water (Jervis 1969). The accumulation of both nutrients and organic matter in the shallow self-contained waters of the prairie potholes contributes to high rates of production. While the estimations of these rates are getting more numerous and precise with time, they are subject to important sources of error. The methods of above-ground biomass estimation consist of using measurements of peak biomass and maximum minus minimum biomass, disappearance methods, multiple harvests or gas exchange (de la Cruz 1978). The most commonly used techniques are the single and multiple harvest methods which underestimate net primary production by as much as 50% in some cases (Whigham *et al*. 1978). Van der Valk and Davis (1978) calculated that, in prairie glacial marshes, the maximum standing crop values underestimated net primary production by 2 to 22% for *Sparganium eurycarpum* and *Carex atherodes*, respectively. These errors originate, for the most part, from ignorance of the importance of leaf mortality, herbivory and the growth of tissues after harvest. Similarly, the estimation of below-ground production is subject to important errors due to difficulties in distinguishing living from dead roots (de la Cruz 1978) or because the pattern of root biomass change has not been well elucidated and is variable between species (van der Valk & Davis 1978).

Table 2.13 lists production values for some wetland components or species. The emergents are much more productive than the submergents. The highest annual production of cattails is three to four times greater than the highest value for submergents. Differences also exist in the contribution of underground biomass to total standing crop: in submergents,

Table 2.13. Annual production and maximum standing crop values for some wetland macrophytes.

Vegetation Component	Annual Production (g/m^2/yr)	Maximum Standing Crop (g/m^2)	Location	Source
EMERGENTS				
1. Reeds				
Phragmites communis	1678 - 2066	654 - 3999	Freshwater tidal wetlands (east coast, U.S.A.)	Whigham *et al*. 1978
P. communis		934	Prairie glacial marshes, Iowa	van der Valk and Davis 1978
2. Rushes				
Scirpus lacustris	1556 ± 684		Freshwater marsh, Utah	Smith and Kadlec 1985
S. maritimus	590 ± 146		"	"
S. fluviatilis	1533		Theresa Marsh, Wisconsin	Klopatek 1975
S. acutus		851	Prairie glacial marshes, Iowa	van der Valk and Davis 1978
S. fluviatilis		466	"	"
S. validus		330	"	"
3. Cattails				
Typha latifolia	2532 ± 948		Freshwater marsh, Utah	Smith and Kadlec 1985
Cattails	1904		Freshwater marsh, New Jersey	Jervis 1969

Table 2.13 (cont'd)

Vegetation Component	Annual Production ($g/m^2/yr$)	Maximum Standing Crop (g/m^2)	Location	Source
T. latifolia	2456		Theresa Marsh, Wisconsin	Klopatek 1975
Typha sp.	956 - 1868	850 - 2338	Freshwater tidal wetlands (east coast, U.S.A.)	Whigham *et al*. 1978
T. glauca		1156	Prairie glacial marshes, Iowa	van der Valk and Davis 1978
4. Sedges				
Sedge swale	1491		Freshwater marsh, New Jersey	Jervis 1969
Sedge shrub	1698		"	"
Carex spp.	900 - 1300		"	Bernard and Gorham 1978
C. lacustris	1186		Theresa Marsh, Wisconsin	Klopatek 1975
Carex spp.		667	Prairie glacial marshes, Iowa	van der Valk and Davis 1978
5. Others				
Phalaris arundinaceae	2028		Theresa Marsh, Wisconsin	Klopatek 1975
Salix interior	1902		"	"
Sagittaria latifolia	1071	214 - 649	Freshwater todal wetlands (east coast, U.S.A.)	Whigham *et al*. 1978

Table 2.13 (cont'd)

Vegetation Component	Annual Production ($g/m^2/yr$)	Maximum Standing Crop (g/m^2)	Location	Source
Sagittaria latifolia		460	Prairie glacial marshes, Iowa	van der Valk and Davis 1978
Sparganium eurycarpum		638	"	"
SUBMERGENTS				
Potamogeton pectinatus		250	Delta Marsh, Manitoba	Anderson 1978
Submergents	100 - 700		Temperate zone	Wetzel 1975
Submergents		192	Prairie glacial marshes, Iowa	van der Valk and Davis 1978

Table 2.14. Primary production of the major ecosystems of the world[a].

		Net Primary Productivity Per Unit Area	
Ecosystem Type	Area	Normal Range	Mean
	10^6 km^2	dry g/m^2/yr	
Tropical forest	24.5	1000-3500	3000
Temperate forest	12.0	600-2500	1250
Boreal forest	12.0	400-2000	800
Woodland and shrubland	8.5	250-1200	700
Savanna	15.0	200-2000	900
Temperate grassland	9.0	200-1500	600
Tundra and alpine	8.0	10- 400	140
Desert and semidesert	42.0	0- 250	40
Cultivated land	14.0	100-3500	650
Swamp and marsh	2.0	800-3500	2000
Lake and stream	2.0	100-1500	250
Total continental	149.0		773
Open ocean	332.0	2- 400	125
Continental shelf, upwelling	27.0	200-1000	360
Algal beds, reefs, estuaries	2.0	500-4000	1800
Total marine	361.0		152
World total	510.0		333

[a] From Whittaker 1975

below-ground biomass contributes less than 10% while in emergents, it may vary between 30 and 70% (Wetzel 1975). Among the emergents, the cattails and reeds are the most productive components with values as high as 2532 and 2066 g/m^2/yr, respectively. This is in agreement with the values reported by Richardson (1979) in his review of primary production in freshwater wetlands. While his average values are higher than those reported in Table 2.13, his ranking of the vegetation by productivity is consistent with the one suggested here: cattails (2740 g/m^2/yr) > reeds (2100 g/m/yr) > sedges (1040 g/m^2/yr). In contrast, Jervis (1969) found very similar net production rates in four communities in a freshwater marsh in New Jersey: 1457 g/m^2/yr for an open aquatic system of emergent and floating plants, 1904 g/m^2/yr for a cattail community, 1491 g/m^2/yr for the sedge-swale system and 1699 g/m^2/yr for the sedge-shrub system. The similarity of the values may reflect the fact that the stands were not monospecific in nature.

The range of values for any one study is usually large and reflects a high level of heterogeneity in productivity both in space and time (Wetzel 1975). The source of this variation comes in part from various forms of biomass loss during growth, such as herbivory or shoot and leaf mortality which are major complicating factors in the estimation of production rates. While Wetzel (1975) reported estimated senescence and death during active growth at 2 to 10% of maximum biomass, respectively, Smith and Kadlec (1985) found that grazing intensity by waterfowl and muskrats varied between 8.9% of above-ground biomass of *Scirpus maritimus* and 47.5% for *Typha latifolia*. Herbivory in the latter study refers not only to the consumption of plant material, but also to the destruction of vegetation for nest building. Leaf mortality can also be important as demonstrated in freshwater tidal wetlands where mortality of tagged leaves on four plant species ranged between 61.3 and 77.7% over a 55-day period (Whigham *et al*. 1978). Shoot mortality may also be high. Bernard and Gorham (1978) showed that 80-90% of emerging shoots in sedge wetlands died before completing their normal life-span. Consideration of life history characteristics of each species is also necessary in the estimation of production in wetlands. For example, ignoring winter green standing crop may lead to overestimation of production.

In spite of the large variability in production rates reported for freshwater wetlands, these values are very high. Jervis (1969) pointed out how marsh flora is particularly well adapted to its environment. The vertical orientation of the leaves avoids intense heating and exposes a maximum of leaf surface. In addition, the ratio of leaf area per unit ground area (leaf area index or LAI) for wetland species is two to three times greater than that of forest or herbaceous communities (Richardson 1979). LAI values for *Typha* are as high as 14.6 while field crop ratios range from two to three (Jervis 1969). Finally, the presence of rhyzomes in any marsh species allows for the mobilization of food to rapidly develop the photosynthetic surface early in the season (Jervis 1969).

The pattern of seasonal biomass change follows a sigmoid curve which decreases later in the season. Growth of four stands of marsh vegetation showed a gradual increase of productivity in spring and a peak in early summer (Jervis 1969). However, the stand of cattails had a high rate in spring due to the mobilization of food stored in the extensive rhyzome system. For most stands, mid-summer productivity was low and there was a net loss in late summer and fall. An exception was the open aquatic community which maintained high mid-summer values and showed net gains in late summer. Jervis (op. cit.) noted that different species carried the production load of the marsh as they went through their respective growth cycles.

In addition to the seasonal cycle of growth, there is a 5-20 year cycle characteristic of some marshes which is related to the water regime. van der Valk and Davis (1978) have described four stages to this cycle. The dry marsh stage is characterized by the germination of seeds of perennial marsh emergents and annual mud-flat species. When flooding takes place, the marsh regenerates as mud-flat species are eliminated, emergent shoot densities increase through vegetative growth and submergent species reappear. This stage is characterized by the dominance of emergents. In the third stage, the degenerating marsh stage, there is a decline of emergents due to the failure of some species to reproduce vegetatively or to the consumption of the plants by muskrats in some instances. Finally, in the lake stage, emergents are almost completely eliminated in favor of submergents and free-floating species. In a study of production in an Iowa marsh during one six-year cycle, annual production changed 18-fold in five years from a peak of 670 tonnes during the height of emergent dominance to a low of 53 tonnes during the lake stage (Weller & Fredrickson 1973).

Although water depth may not be the only factor causing changes in a marsh community as described above, it does affect production. Lathwell *et al*. (1973) found that, in a marsh with a water depth of 30.5 cm, *Scirpus* and *Chara* constituted 73 and 18%, respectively, of a total standing crop of 8.5 tonnes/ha while at a depth of 91.4 cm their contributions had changed to 17% *Scirpus* and 64% *Chara* of a total standing crop of 3.3 tonnes/ha. Therefore, increased water levels reduced the productivity of both species in spite of the increased contribution of the submergent *Chara*.

The production of algae relative to that of macrophytes cannot be determined for wetlands as both are rarely measured simultaneously. Also, differences in measurement techniques make comparisons difficult: macrophyte production is evaluated as biomass produced and algal productivity is most often measured as the amount of carbon assimilated in a fraction of time. Wetzel (1975) reported epiphytic production greater than 20% of total production in the littoral zone of certain lakes, exceeding the contribution of phytoplankton. This underlines the importance of nutrient exchanges between macrophytes and attached algae, which give the epiphytes a competitive edge over the phytoplankton. There is a large amount of variation in the production of epiphytic algae between and within habitats.

The production of algae growing on Scirpus can vary four-fold between habitats and a similar variability in productivity exists between algae growing on different macrophyte species: rates of production per m^2 of plant surface area were four times greater on Scirpus than on Potamogeton (Table 2.15). However, because of the larger surface area per m^2 of macrophyte zone offered by Potamogeton and other submergents, production of epiphytic algae tend to be much higher in beds of submergents than in areas of emergent growth (Table 2.15).

2.4 The Invertebrates of Prairie Sloughs

The combination of a high level of primary production, a rich flora and a structurally complex and variable vegetation, along with fluctuating environmental conditions, contribute to high levels of microhabitat diversity in prairie potholes. This diversity, in turn, favors the development of a very rich invertebrate fauna. Table 2.16 lists the invertebrate taxa sampled in prairie potholes. While this list covers 10 phyla and 38 orders, suborders or families, it is in no way exhaustive. The higher level of detail in the taxonomic characterization of the class Insecta reflects more the research interests of the workers than the taxonomic dominance of that class. For example, while Voigts (1976) sampled a maximum of 32 invertebrate taxa in Iowa marshes during a study of invertebrate-vegetation relationships, Driver (1977) found as many as 27 species of chironomid larvae in one pond in Saskatchewan when focusing solely on the Chironomidae. Therefore, it is clear that the taxonomic richness reported in various studies is heavily biased by the purpose of the work carried out. For this reason it is difficult to get an estimate of invertebrate species richness in the sloughs. Most, if not all, investigations looking at invertebrates in prairie potholes do so because of their importance as duck foods. Thus, the focus of the attention of the researchers is heavily bent towards the classes Gastropoda, Crustacea and Insecta because they are the preferred invertebrate foods of ducks (see Section 2.5). These three classes are from only two of the ten phyla found in Table 2.16 and probably constitute only a small fraction of the total number of species present.

In the study of wetland invertebrates, bias is also introduced through the sampling techniques used. The methods or tools, most often cited are: the dip net, the tube sampler, the Ekman dredge, the stove pipe sampler, the activity trap, and the removal of individual plants through enclosure in plastic bags. All the above differ in the habitat they were designed to sample and in their ability to estimate absolute densities. The dip net and tube sampler are best suited for the sampling of open water areas although the net ignores small specimens and is not 100% efficient for the larger and more mobile individuals. The Ekman dredge is effective in sampling the sediments but may become less efficient in weedy areas. In densely vegetated habitats, the stove pipe sampler may be the most thorough method because the water, bottom and vegetation can all be

Table 2.15. Primary production of epiphytic algae on macrophytes.

Site	Production per m^2 of Plant Surface Area ($g\ C\ m^{-2}$)	Production per m^2 of Macrophyte Zone ($g\ C\ m^{-2}$)	Location	Source
Scirpus	32.3	43.5	Crescent Pond, Delta Marsh, Manitoba	Hooper and Robinson 1976
Typha				
High density	23.0	22.9	"	"
Low density	28.4	5.7	"	"
Phragmites	2.3	0.6	"	"
Potamogeton				
High density	8.6	48.5	"	"
Low density	7.8	11.8	"	"
Scirpus	122.6	71.5	Lawrence Lake, Michigan	Allen 1971
Najas - *Chara*	94.2	659.4		

Table 2.16. Invertebrate taxa found in prairie sloughs[a].

Phylum	Class (subclass)	Order (suborder)	Family	Common Name
Protozoa				Protozoans
Porifera	Desmospongea	Haplosclerina	Spongillidae	Fresh water sponges
Coelenterata	Hydrozoa			Hydras
Platyhelminthes	Turbellaria			Flatworms
Gastrotricha	Chaetonotoidea			Gastrotrichs
Rotatoria				Rotifers
Nematoda				Nematodes
Annelida	Oligochaeta			Aquatic earthworms
	Hirundinea			Leeches
Mollusca	Gastropoda	Basommatophora	Physidae	Fresh water snails
			Lymnaeidae	(e.g., *Lymnae*, *Physa*,
			Planorbidae	*Helisoma*)
	Pelecypoda		Unionidae	Freshwater mussels
			Sphaeriidae	Freshwater clams
Arthropoda	Arachnida	Acarina		
		Trombidiformes		Water mites
	Crustacea			
	Branchiopoda	Anostraca		Fairy shrimps
		Notostraca		Tadpole shrimps
		Diplostraca		
		Conchostraca		Clam shrimps
		Cladocer		Water shrimp
	Ostracoda			Seed shrimp
	Copepoda			Copepods
	Malacostraca	Isopoda		Isopods (aquatic sow bugs)
		Amphipoda		Amphipods (scuds, side swimmers)

Table 2.16 (cont'd)

Phylum	Class (subclass)	Order (suborder)	Family	Common Name
Arthropoda (cont'd)	Insecta	Collembola		Springtails
		Ephemeroptera		Mayflies
		Odonata		
		Anisoptera		Dragonflies
		Zygoptera		Damselflies
		Hemiptera	Gerridae	Water Striders
			Notonectidae	Back Swimmers
			Belostomatidae	Giant water bugs
			Corixidae	Water boatmen
		Neuroptera		Dobsonflies
		Trichoptera		Caddisflies
		Coleoptera		Aquatic beetles
			Dystricidae	Predaceous diving beetles
		Diptera		Flies, mosquitoes, midges
			Culicidae	Mosquitoes, phantom midges
			Chironomidae	True midges
			Stratiomyidae	Soldier flies

---

[a] Taken mostly from Perret 1962, Sugden 1964 and Swanson *et al*. 1974.

removed and screened for invertebrates with a high level of accuracy. Activity traps are relatively easy to use with little effort and time in all habitats, thus allowing large numbers of samples to be taken; unfortunately, they only provide relative estimates of abundance and select for highly mobile species. Finally, in the study of the association between species of plants and invertebrates, the use of plastic bags to enclose and remove individual plants has probably been the most precise technique.

In light of the variety of methods used to sample invertebrates in wetlands, comparisons of diversity and abundance between studies is very difficult. The differences in techniques used and, consequently, the lack of comparable data sets reflect more the differences in the aims of the studies than the lack of adequate sampling methods. However, the thorough investigation of all invertebrates would be a very complex and time-consuming enterprise. In spite of the difficulties involved in the study of wetland invertebrates, it is clear, for the reasons cited at the beginning of this section, that the fauna of prairie sloughs is quite diverse due to the variety of habitats offered in these systems both in time and in space.

While lower invertebrates certainly play an important role in wetland dynamics, the following discussion of the distribution and functional aspects of invertebrates in prairie sloughs will focus mostly on the phyla Annelida, Mollusca and Arthropoda, as the others have been mostly ignored in the studies of wetlands. Microorganisms, because of the key role they play in decomposition processes, will be discussed seperately in the subsection on functional relationships.

2.4.1 Invertebrate abundance and distribution

The estimation of invertebrate production in aquatic systems is complex because it requires the measurement not only of standing crop but also of the rates of biomass turnover. This, in turn, requires the determination of the growth curves and the number of generations for each species which can be effectively measured for only a few dominant species because of the extensive sampling required. Invertebrate production estimates in wetlands are therefore rarely available. However, the positive relationship between benthic invertebrate biomass and the general productivity of lakes (Wetzel 1975) suggests that potholes may have high levels of invertebrate production due to their characteristically high rates of primary production (see Section 2.3.3).

There are three distinct habitats within wetlands for invertebrates: the bottom, the open water and the vegetation. Of these three, the vegetation harbors the largest number of invertebrates.

Macroinvertebrate standing crops in beds of macrophytes are more than twice the average for whole lake systems (Table 2.17). This difference probably reflects the importance of the vertical component provided to the structure of the habitat by dead and living aquatic plants which contribute a greater surface area for invertebrate colonization and feeding (see review by Moyle 1961). The surface area of macrophytes can be as large as 50 times that provided by the substrate above which it is found (Edwards & Owens 1965, cited in Hurlbert 1975). Joyner (1982) found on average over 5 and 20 times more individuals in samples from the vegetation of a marsh than in samples from the substrate and the water column, respectively. The size of the differences may be exagerated as the zooplankton were not sampled by the methods used. Densities of zooplankton can be very high at certain times: Collias and Collias (1963) recorded over 1000 Daphnia per litre in open water areas of Delta Marsh. In Lawrence Lake, Michigan, invertebrate biomass was 1.4 to 5.1 times greater in the vegetation zone than on bare sediments (Mittelbach 1981). While benthic invertebrates have been shown to be several times more numerous in vegetated than non-vegetated areas (Krull 1970), numbers on the foliage of submergent vegetation were similar to those on the sediments beneath the same plants in Lake Opinicon, Ontario (Keast 1984).

The taxa of invertebrates most often cited as dominant in freshwater marshes or wetlands are the gastropods, amphipods, notonectids, corixids and the larvae of chironomids, trichoptenans, odonates and ephemeropterans (Rosine 1955, Dvorak 1970b, Krull 1970, Joyner 1980, Mittelbach 1981, Biggs & Malthus 1982, Dvorak & Best 1982, Joyner 1982). The gastropods, while not the most numerous, often contribute the most to total invertebrate biomass. Although sometimes associated with the substrate, their densities increase with increasing vegetative cover. They are closely associated with dense stands of both emergent and submergent plants and also with floating species. Amphipods, can be very abundant in association with submergent vegetation and under floating species such as the duckweed Lemna. Larval chironomids are common to both the bottom and submerged vegetation; large numbers may also be associated with patches of blue-green algae (Collias and Collias 1963). The larvae of both trichopterans and ephemeropterans are associated with a variety of substrates such as rocks, sand, bottom mud, vegetation or detritus. The naiads of dragonflies and damselflies are usually found on the submerged parts of emergent plants. The notonectids and corixids are usually sampled with dip nets in the water column within vegetated areas and zones of open water. As mentioned earlier, some zooplankton can become very dense in the water column of weed-free areas where phytoplankton blooms can occur.

2.4.1.1 Plant-invertebrate associations

In light of the importance of vegetation to invertebrate distribution and abundance, understanding the association between macrophytes and invertebrates is of great importance as most of the

Table 2.17. Invertebrate standing crop values for some ecosystems and habitats.

Invertebrates and Habitat	Wet Weight Biomass (gm^{-2})	Source
Benthic macroinvertebrates (excluding molluscs)		extracted from table 16-3 of Wetzel 1975
38 lakes of USA mostly Connecticut and New York	1.0 - 34.8	
13 lakes of northern Canada	8.9	
64 lakes of northern Germany	11.5	
15 pond-type Polish lakes	2.0 - 37.0	
shallow fish culture ponds Czechoslovakia	22.2 - 27.2	
Dominant benthic invertebrates Lake Borrevann, southern Norway		Okland 1964
0.2 m	17.9	
1.5 m	45.8	
Benthic invertebrates, Lake Opinicon, Ontario		Keast 1984
weed beds at 1.0-1.3 m	42.1 - 106.5	
2.5 m	10.1 - 32.2	
Invertebrates of macrophyte zone, Lake Vechten, Netherlands	50[a]	Dvorak and Best 1982

Table 2.17 (cont'd)

Invertebrates and Habitat	Wet Weight Biomass (gm^{-2})	Source
Invertebrates of *Scirpus* beds, Lawrence Lake, Michigan	8.0 - 14.7[a]	Mittelbach 1981
Macroinvertebrates of *Carex*-*Glyceria* stand, Radov Pond, Czechoslovakia	30.0 - 80.0	Polunin 1984
Macroinvertebrates, Radov Pond, Czechoslovakia		Dvorak 1978
Glyceria-*Carex*	average 42.18 maximum 65.0	
Sparganium-*Glyceria*	average 66.04 maximum 72.6	
Benthic invertebrates, experimental ponds	maximum 52.0[a]	Hall *et al*. 1970

[a] Values obtained by multiplying dry weight by 10

smaller potholes in the prairies are covered by an abundant and diverse marsh vegetation.

Perret (1962) sampled the fauna of a number of potholes in the Minnedosa district of Manitoba and found that the samples from the edges of the potholes, dominated by emergents, and those from the centres which consisted mainly of submergents were very similar in total abundance and taxomomic composition of invertebrates. However, the relative abundance of the invertebrate species sampled were different between the two sites. The amphipods and dipterans constituted 14.1% and 78.2%, respectively, of the invertebrates sampled in the centre of the potholes while on the edge the proportions were 40.1% and 47.0%. Sampling of macroinvertebrates associated with different vegetation types in a ditch demonstrated the preference of Coleoptera for areas of emergents and the preference of Hemiptera and Diptera for zones dominated by submergents (Scheffer et al. 1984). A two year study in four Iowa prairie marshes noted that the four most abundant groups of organisms were closely associated with the vegetation (Voigts 1976). The isopods were most abundant within stands of emergents and floating dead vegetation while the amphipods and chironomids dominated areas of dense submergent growth. Gastropods were most abundant within dense stands of emergents or areas of dense submergents and free-floating vegetation. These plant-invertebrate associations were strong: changes in the abundance of the dominant fauna over time were closely tied to changes in their preferred habitat.

A more detailed study of plant-invertebrate associations was done in the littoral zone of Lake Vechten, The Netherlands (Dvorak & Best 1982). Eight plant species were sampled for invertebrates: three emergent species, four submergents and one floating-leaved species. The authors found that all eight species of macrophytes had many invertebrate species in common although a few were exclusively associated with the emergents and the floating-leaved species. The difference between the fauna of submergents and that of emergents found in other studies was not as big in this study as the fauna of the emergent Sparganium more closely resembled that of Myriophyllum, a submergent. Also, among the four species of submergents, rooting depth played a more important role in determining faunal composition than did species-specific differences. Similarly, Gerrish and Bristow (1979) found no marked difference in the faunal composition of three species of submergents. Macrophytes sampled in streams also showed little difference in the kinds of invertebrates they harboured (Rooke 1984). Rosine (1955) found that in Muskee Lake, Colorado, nearly all species of invertebrates occured to some extent on each of three submerged aquatic plants, although some showed clear preferences. In contrast to these studies, Krull (1970) found strong species-specific associations between invertebrates and 12 species of submerged aquatic plants which were sampled weekly in five marsh sites within the Montezuma National Wildlife Refuge, New York, over one season. Of 79 taxa of aquatic invertebrates sampled with the plants, 52 (66%) were found associated with only three or fewer plant species and 33 (42%) were found on only one plant species. The larger number of species-specific associations recorded in this study probably reflects the variety of habitats sampled, the larger number of plant species sampled and the more extensive sampling regime. These factors are reflected in the diversity of species of invertebrates sampled: 114 taxonomic groups were recorded.

Between site differences in relative abundance are also found within monospecific stands of vegetation. Dvorak (1970a) found that, in a stand of the emergent Glyceria aquatica, the relative contribution of the most abundant invertebrates changed rapidly along a transect from the edge of the pond (water depth = 10 cm) to the edge of the open water zone (water depth = 50 cm). The contribution of molluscs decreased while that of the culicids increased with distance from shore. The last sampling station, closest to the open water zone, was strikingly different from all others: leeches and hemipterans clearly dominated.

Aquatic plants not only provide a substrate for invertebrate colonization but also modify their environment thereby influencing invertebrate species composition. For instance, certain chemical and physical properties of the water within a monospecific stand of Glyceria aquatica changed rapidly as one moved from the edge towards the open water zone of the pond: the temperature, alkalinity, pH and concentration of dissolved oxygen were all higher in water sampled closest to the open area (Dvorak 1970a). These changes were correlated with changes in the dominant invertebrates. Keast (1984), who found that benthic invertebrates were over three times more numerous under beds of Potamogeton and Vallisneria than under Myriophyllum, explained that the observed differences were due to the differences in the nature of the substrate upon which the plants grew. Myriophyllum spicatum forms dense belts from bottom to surface which slow down water movement and trap the finer material. The loose sediments thus formed support fewer invertebrates.

The abundance of invertebrates also varies with aquatic plant species. Table 2.18 shows some values of invertebrate abundance within stands of macrophytes. There are fairly large differences between studies in the values reported due mostly to differences in the ecosystems sampled and the sampling techniques used. In spite of this variability, it is clear that certain species of plants tend to support greater invertebrate populations. While it is hard to discern trends in light of the small data base, some workers have suggested that plant species with more finely divided leaves support a more abundant fauna (Rosine 1955, Krull 1970, Gerrish & Bristow 1979). Gerrish and Bristow (1979) found that artificial plants modeled on Potamogeton harboured as many invertebrates as the living plants, thus suggesting that form played an important role in determining faunal abundance. Factors other than form may be important as Korinkova (1971) did not find a relationship between leaf dissection and animal abundance in three species of submerged macrophytes. For instance, Macan and Kitching (1972) demonstrated how invertebrate abundance was correlated with vegetation density. They found that the numbers of invertebrates

Table 2.18. Abundance of invertebrates associated with various aquatic plant species. Values are means except where indicated.

Species	Ind/m^2	Ind/g Dry Weight of Plant	Ind/m^2 Surface Area of Plant	Site	Source
Potamogeton cheesemanii	232			Lakes, New Zealand	Biggs and Malthus 1982
Ranunculus fluitans	512			"	"
Myriophyllum propinquum	1202			"	"
Elodea canadensis	1362			"	"
Characea (*Chara* & *Nitella*)	4958			"	"
Potamogeton natans	788			Ditch, The Netherlands	Scheffer *et al*. 1984
Stratoides aloides	1136			"	"
Sium erectum	1796			"	"
Elodea canadensis & *Ranunculus circinatus*	1308			"	"
Potamogeton lucens & *Elodea canadensis*	4980 6240	77 37	919 461	Carp pond, Czechoslovakia	Korinkova 1971
Potamogeton robbinsii	7640[a]	38[a]		Lake Opinicon, Ontario	Keast 1984
Vallisnera americana	4560	40		"	"
Myriophyllum spicatum	5499	90		"	"

Table 2.18 (cont'd)

Species	Ind/m^2	Ind/g Dry Weight of Plant	Ind/m^2 Surface Area of Plant	Site	Source
Ceratophyllum demersum		1.6		Marshes, Montezuma National Wildlife Refuge, New York	Krull 1970
Potamogeton pectinatus		1.0			"
P. foliosus		0.66		"	"
Lemna trisulca		1.5		"	"
Chara vulgaris		0.44		"	"
Elodea canadensis		0.6		"	"
Najas flexilis		0.4		"	"
Myriophyllum spicatum		0.4		"	"
Potamogeton richardsonii			7862^a	Lake Opinicon, Ontario	Gerrish and Bristow 1979
Vallisneria americana			2045	"	"
Myriophyllum exalbescens			20000	"	"
Polygonum natans			800	Muskee Lake, Colorado	Rosine 1955
Chara deliculata			600	"	"
Potamogeton gramineus			200	"	"

a Values reported for this study are the highest recorded colonizing artificial substrates made of strands of rope increased with greater densities of strands. Therefore, leaf dissection and density of the weed beds seem to interact in determining invertebrate abundance.

While plant morphology and growth form affect invertebrate numbers on plants, vegetative diversity within wetlands is important in determining the level of invertebrate production. Voigts (1976) found in Iowa prairie marshes that "although the open submergent phase produced the most invertebrates, it is important that this phase be interspersed with the emergent phase for maximum production".

2.4.1.2 Variation between potholes

Few studies have looked at the variance in invertebrate community composition and abundance between potholes. Perret (1962) sampled the fauna of more than 50 potholes over three years but did not report the data in a fashion which allows for comparison of abundance between ponds. However, the frequency of occurrence of certain taxa among the potholes is presented. The distribution of the fauna inhabiting the edge of the potholes was variable as all groups of organisms were not represented in all ponds: while dipterans were present in almost 100% of the potholes, ephemeropterans and hemipterans were found in 50% or less of the ponds. Bartonek and Hickey (1969b) sampled five potholes with a net and dredge at two-week intervals over two summers. They found that the taxa of invertebrates sampled and their abundance "varied markedly among the five potholes as well as within potholes." It seems, from these two studies, that variability in invertebrate abundance between potholes may be as high as that found within potholes.

2.4.1.3 Variability between years

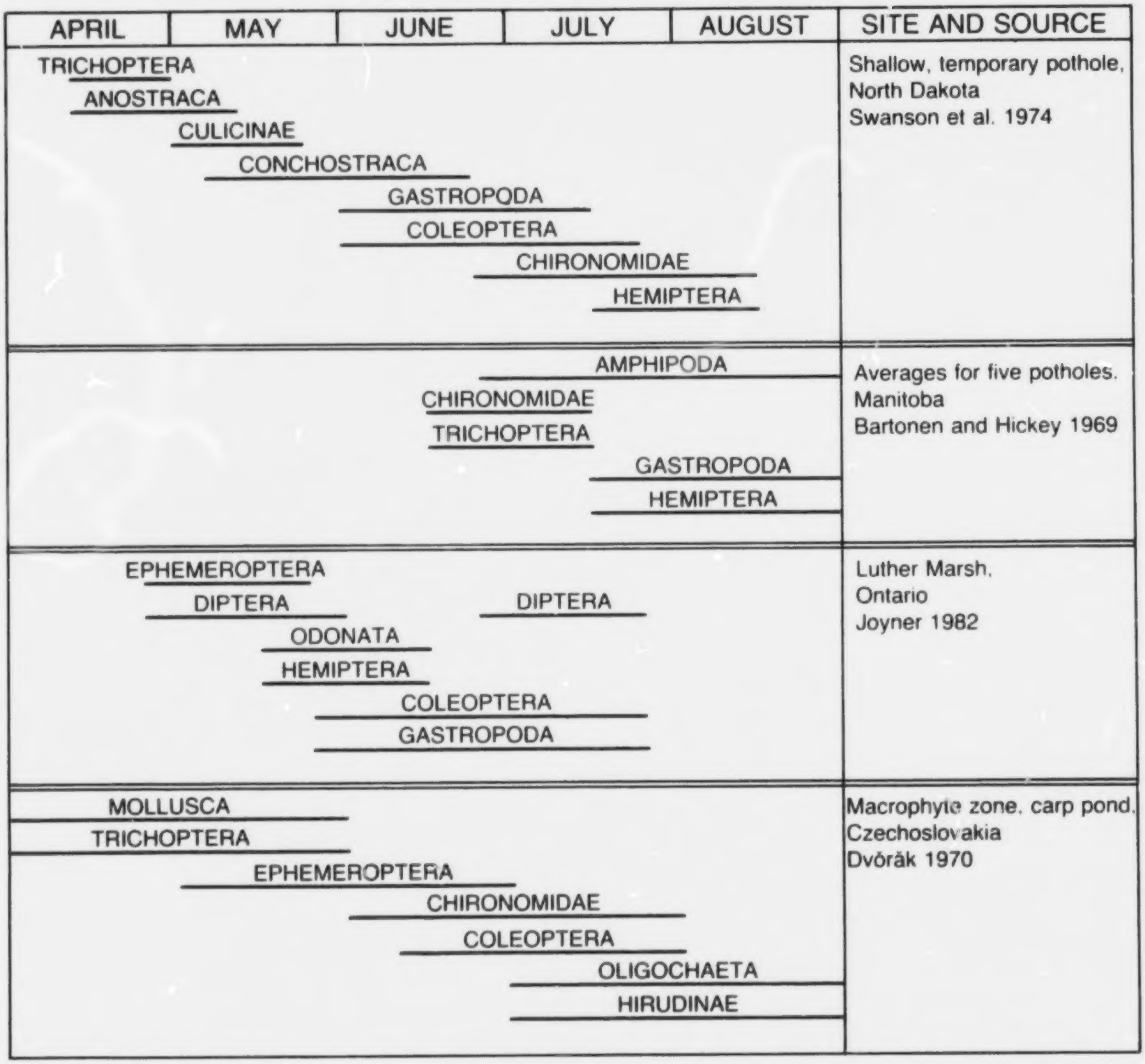
In five potholes of the Minnedosa area of Manitoba, differences in abundance between two years were large for cladocerans, gastropods and especially amphipods whose numbers went from representing 71% of the invertebrates sampled in one pothole to 0% the next year (Bartonek & Hickey 1969b). Voigts (1976) found significant changes in the abundance of certain taxa over two years in prairie marshes. These changes were more pronounced for amphipods than for other invertebrates. Perret (1962), sampling a large number of potholes over three years in Manitoba, found that while the total abundance of bottom fauna did not change significantly between years, changes in certain taxa were great: numbers of Ephemeroptera, Trichoptera, Gastropoda and Amphipoda fluctuated substantially between years, with amphipod densities changing eightfold between 1957 and 1958. Also, the frequency of occurrence among potholes changed drastically for certain taxa, indicating the total elimination of some species in a number of ponds. Amphipods were found in over 75% of the potholes sampled in 1958 but only in less than 25% in 1957. The frequency of occurrence of leeches dropped from 70 to 25% in the same time span. The variability between years reported in two of the studies (Perret 1962, Voigts 1976) were both related to changes in water regime and, consequently, in vegetation patterns.

2.4.1.4 Seasonal patterns of abundance

Few studies have been conducted on the seasonal dynamics of pothole invertebrates. However, some general trends can be described. Total invertebrate abundance and/or biomass usually reached its highest values in late spring or early summer (late May to late June) in a number of wetlands studied (Dvorak 1970b, Krull 1970, Swanson et al. 1974, Gerrish & Bristow 1979, Mittelbach 1981, Joyner 1982). In contrast, Perret (1962) found that the bottom fauna usually increased in abundance to a spring peak followed by a slight depression and then a large increase until a summer peak was reached. The results obtained by Perret may differ because of the bias introduced by sampling only the bottom of the potholes. It has been noted that the chief maxima in abundance and biomass tend to occur during the peak development of the vegetation (Dvorak 1970b, Krull 1970). Pip and Stewart (1976) showed that the association between two species of snails and their preferred plant species was strongest during periods of active growth of the plants.

Seasonal changes in faunal abundance are due, for the most part, to the succession of individual species of invertebrates (Dvorak 1970b). Seasonal patterns of abundance for individual taxa are highly variable. Figure 2.6b shows the periods of the year during which the dominant taxa are at their most abundant in four different habitats. In light of the fact that the four studies presented were carried out in different habitats at different times, it is not surprising to find widely diverging patterns of abundance. Little can be said about species succession in potholes due to the lack of adequate data. Although two of the studies presented in Figure 2.6b were carried out in prairie potholes, one presented data from temporary sloughs (Swanson et al. 1974) and the other from more permanent and deeper ponds (Bartonek & Hickey 1969b). The temporary ponds were characterized by wet meadow and shallow marsh vegetation. The early dominant invertebrates were anostracans, cladocerans and conchostracans, all typical fauna of shallow temporary bodies of water. Dipterans were common throughout the season with culicids important early in the spring and chironomids dominating in the second half of the season. Gastropods and coleopterans contributed significantly to the fauna in June and July while hemipterans were important late in the season before the potholes went dry. Bartonek and Hickey (1969b) found that in more permanent potholes the fauna was dominated by amphipods, cladocerans, chironomids and gastropods. While trichopterans and chironomids were more important in June, the corixids and gastropods were more numerous later in August. Although amphipods and cladocerans tended to reach their maxima in early July, they were quite abundant throughout most of the summer.

Figure 2.6b. Periods of greatest abundance of dominant invertebrate taxa in four wetland ecosystems.


APRIL
MAY
JUNE
JULY
AUGUST
SITE AND SOURCE
TRICHOPTERA
ANOSTRACA
CULICINAE
CONCHOSTRACA
GASTROPODA
COLEOPTERA
CHIRONOMIDAE
HEMIPTERA
Shallow, temporary pothole, North Dakota Swanson et al. 1974
AMPHIPODA
CHIRONOMIDAE
TRICHOPTERA
GASTROPODA
HEMIPTERA
Averages for five potholes, Manitoba Bartonen and Hickey 1969
EPHEMEROPTERA
DIPTERA
DIPTERA
ODONATA
HEMIPTERA
COLEOPTERA
GASTROPODA
Luther Marsh, Ontario Joyner 1982
MOLLUSCA
TRICHOPTERA
EPHEMEROPTERA
CHIRONOMIDAE
COLEOPTERA
OLIGOCHAETA
HIRUDINAE
Macrophyte zone, carp pond, Czechoslovakia Dvǒrák 1970

Both Mittelbach (1981) and Armstrong and Nudds (1985) found a decrease in the size of invertebrates as the season progressed. This shift in size corresponded to the loss of larger instars of Ephemoptera, Trichoptera, Odonata and Amphipoda (Mittelbach 1981).

2.4.2 Factors affecting invertebrate assemblages

2.4.2.1 Water regime

The results presented above on seasonal patterns of invertebrate abundance in prairie potholes clearly show how the dominant invertebrate taxa change with pond type. Pond permanency, vegetative structure and water regime are all closely related thereby suggesting that water regime may influence the invertebrates of potholes. Driver (1977), looking at chironomid communities of small prairie potholes, found such a relationship. Sixteen ponds were sampled for chironomids: ten temporary, five semi-permanent and one permanent pond. The temporary ponds held water for less than a year while semi-permanent and permanent potholes held water for at least 2 and 25 years, respectively. Driver found that among the temporary ponds, chironomid diversity increased with the length of time the pond held water. The highest measures of diversity were recorded in semi-permanent potholes. While temporary ponds were usually dominated by one or two species, the semi-permanent and permanent potholes tended to have more even abundance patterns. Driver concluded that chironomid diversity was dependent on the stage of development of the plant community which was, in turn, affected by the moisture regime. Temporary ponds have a simple plant community which does not offer much diversity in species or structure. In contrast, more permanent ponds are characterized by a mixture of emergent and submergent vegetation which provides heterogeneity in the environment thereby favoring more complex chironomid communities.

Changes in water levels can modify the plant environment and consequently the invertebrate community. For example, increased water levels in one pothole over three years resulted in a reduction in abundance of the vegetation and in the number of plant species; in the third year the number of chironomid species dropped from 22 to 10 (Driver 1977). Perret (1962) found that, as heavy emergent plant cover was replaced by open water over two years, the Odonata and Trichoptera decreased in abundance within ponds and occured less frequently among the potholes sampled. The drawdown of a waterfowl impoundment over one summer led to a large drop in invertebrate numbers the following year: Tricoptera, Ephemeroptera and Odonata were almost completely eliminated and dipteran numbers dropped by 60 to 95% (Kadlec 1962). Filion (1967) noted that, in three mountain reservoirs in Alberta, the densities of benthic invertebrates were significantly lower in zones periodically subjected to drawdowns. The drainage of a reservoir in Ontario and the subsequent exposure to freezing temperatures resulted in the loss of the olegochaetes, nematodes, mites and a large percentage of the chironomids (Paterson & Fernando 1969).

2.4.2.2 Salinity

Chironomid species composition does not seem to be affected by lower concentrations of salinity (Driver 1977). Dominant invertebrates in semi-permanent wetlands do change, however, with increasing salinity: between 3,000 and 30,000 mgL^{-1} of salts, amphipods and snails were replaced by anostracans as the dominant species (Swanson et al. 1974). As levels approached 30,000 mgL^{-1}, the brine shrimp, Artemia salina, dominated the fauna. Hartland-Rowe (1966) found that in temporary pools in Alberta and Saskatchewan the number of species within the taxa Anostraca which were tolerant of a certain salinity range diminished with increasing salinity.

2.4.2.3 Agriculture

Swanson and colleagues (1974) found that the standing crop of invertebrates within temporary wetlands was dependent on land-use practises. Potholes within agricultural areas had lower seasonal averages of invertebrates than potholes within undisturbed areas. Ponds surrounded by land in summer fallow had the lowest average invertebrate density. These findings may reflect the dependence of small temporary potholes on an allochtonous organic food base. Dwyer (1970), comparing potholes within and outside an agricultural area in Manitoba, recorded larger numbers of free-swimming invertebrates in the agricultural land potholes. The larger amounts of undecomposed organic matter on the bottom of potholes with wooded shores did not allow the establishment of dense submergent vegetation, thereby depriving invertebrates of an important habitat.

2.4.3 Trophic relationships

The aim of this section is to describe the effects of invertebrate activity on pond community metabolism. The rich and productive flora of potholes provides a variety of habitats for an equally rich and abundant fauna as described in the preceeding sections. The life habits and feeding activities of a dense invertebrate assemblage modifies the habitat to some extent.

2.4.3.1 Physical and chemical effects

Bioturbation is an important factor affecting sediments in aquatic systems. Organisms living in or on the bottom can modify the physical and chemical properties of sediments. For example, deposit feeders can modify the size of sediment particles and the chemical nature of the sediments through feeding (Polunin 1984). The burrowing activities of oligochaetes, ephemeropterans or dipteran larvae mix the sediments, thereby enhancing the chemical interchange between the sediments and the overlying layer of water (Davis 1974, Hobbie et al. 1980). These activities of the invertebrates, however, may be of little importance in prairie sloughs in comparison to the turbulence caused by ducks searching for food within the sediments.

The migration of animals in and out of the potholes may influence nutrient dynamics within the systems. Vallentyne (1952) measured an average daily emergence of 136 insects/m^2 over Winona Lake, Indiana; this emergence rate corresponds to a daily loss of 2.26 and 0.15 mg of nitrogen and phosphorus, respectively. These losses could be considerable in a marsh where chironomids are very abundant, especially if they are not replaced through immigration (Kadlec 1979).

2.4.3.2 Foods of invertebrates

The feeding mechanisms and food preferences of aquatic invertebrates determine the trophic relationships within the ecosystem and, consequently, the major routes of energy flow and nutrient recycling. The feeding activities also work indirectly to modify the physical environment. For instance, zooplankton grazing increases transparency thereby stimulating macrophyte growth. The breakdown of dead plant material associated with the feeding habits of certain invertebrates decreases the amount of available substrate within the habitat.

Information on feeding methods and preferences for aquatic insects and other important invertebrate taxa is presented in Tables 2.19 and 2.20, respectively. The feeding mechanisms are varied and each is suited to the dominant food consumed and the habitat selected by each taxa. For example, the Gastropoda, and certain species of Ephemeroptera, Diptera and Trichoptera are associated with the surface of various substrates and feed by scraping. The planktonic forms filter particulates out of the water while inhabitants of the bottom such as oligochaetes, ostracods, amphipods and many dipterans feed on or in the sediments by filtering or simply ingesting the sediments. The predators either swallow their prey whole, as do the odonates and some species of leeches, or they pierce and consume the cell and tissue fluids of the prey, as do some hemipterans or dipterans. One last feeding mechanism which is used by species of Trichoptera, Coleoptera and Diptera is the shredding, either by chewing or mining, of living or decomposing plant tissue.

It is of interest to note that over evolutionary time the aquatic insects have radiated widely within the freshwater aquatic ecosystem to take advantage of the variety of habitats and foods available. Except for the Odonata, all the orders of insects are associated with all four of the major feeding categories (see Table 2.19).

The distribution of aquatic invertebrates by trophic category within wetlands indicates that most of the collectors and scrapers are common to most plants and substrates (Dvorak & Best 1982). Rooke (1984) found that in one stream the scrapers were more closely associated with plants which had more detritus on them. The number of invertebrates which mine plants was highest on emergents and Polygonum in a Dutch lake (Dvorak & Best 1982). Dvorak (1970a) found that the biomass of carnivores in a

Table 2.19. Feeding mechanism and food types of aquatic insects important in potholes[a].

General Category Based on Feeding Mechanism	General Particle Size Range of Food (μm)	Subdivision Based on Feeding Mechanism	Subdivision Based on Dominant Food	Aquatic Insect Taxa Containing Predominant Examples
Shredders	$>10^3$	Chewers and miners	Herbivores: living vascular plant tissue	Trichoptera (Phryganeida, Leptoceridae)
				Coleoptera (Chrysomelidae) Diptera (Chironomidae, Ephydridae)
		Chewers and miners	Detritivores (large particle detritivores): decomposing vascular plant tissue	Trichoptera (Limnephilidae, Lepidostomatidae) Diptera (Tipulidae, Tipulidae, Chironomidae)
Collectors	$<10^3$	Filter or suspension feeders	Herbivore-detritivores: living alga cells, decomposing particulate organic matter	Ephemeroptera (Siphlonuridae) Trichoptera (Philopotamidae, Psychomyiidae, Hydropsychidae, Brachycentridae)
				Diptera (Simuliidae, Chironomidae, Culicidae)
		Sediment or deposit (surface) feeders	Detritivores (fine particle detritivores): decomposing organic particulate matter	Ephemeroptera (Caenidae, Ephemeridae, Leptophlebiidae, Baetidae, Ephemerellidae, Heptageniidae) Hemiptera (Gerridae) Coleoptera (Hydrophilidae) Diptera (Chironomidae, Ceratopogonidae)

Table 2.19 (cont'd)

General Category Based on Feeding Mechanism	General Particle Size Range of Food (µm)	Subdivision Based On Feeding Mechanism	Subdivision Based on Dominant Food	Aquatic Insect Taxa Containing Predominant Examples
Scrapers	$<10^3$	Mineral scrapers	Herbivores: algae and associated micro-flora attached to living and non-living substrates	Ephemeroptera (Heptageniidae, Baetidae, Ephemerellidae) Trichoptera (Glossosomatidae) Helicopsychidae, Molannidae, Odontoceridae, Goreridae)
				Coleoptera (Elmidae, Psephenidae) Diptera (Chironomidae, Tabanidae)
		Organic scrapers	Herbivores: algae and associated attached micro-flora	Ephemeroptera (Caenidae, Leptophlebiidae, Heptageniidae, Baetidae) Hemiptera (Corixidae) Trichoptera (Leptoceridae) Diptera (Chironomidae)
Predators	$>10^3$	Swallowers	Carnivores: whole animals (or parts)	Odonata Megaloptera Trichoptera (Rhyacophilidae, Polycentropidae, Hydropsychidae) Coleoptera (Dytiscidae, Gyrinnidae) Diptera (Chironomidae)
		Piercers	Carnivores: cell and tissue fluids	Hemiptera (Belastomatidae, Nepidae, Notonectidae, Naucoridae) Diptera (Rhagionidae)

a Modified from Cummins 1973

Table 2.20. Feeding mechanisms and food types for invertebrate taxa important to potholes[a].

Taxa	Feeding Mechanism	Dominant Food
Oligochaetes	sediment feeders	decomposing organic particulate matter, algae, bacteria
Hirudinae	ectoparasites	blood and body fluid of vetebrates
	carnivores	invertebrates
Gastropods	scrapers	algae and microflora attached to living (macrophytes) and non-living substrates; also scavenges dead animal material
Cladocera	suspension feeders	algae, bacteria, protozoa, organic detritus
Copepods	suspension feeders and/or raptorial feeders	algae, small crustaeceans organic debris
Ostracoda	sediment feeders	algae, bacteria, detritus
Amphipoda	sediment feeders	bacteria, algae, particulate detritus of plant and animal remains

[a] Information collected from Pennak 1978 and Wetzel 1975

stand of Glyceria aquatica never exceeded 18% of total biomass except at one station at the edge of open water where the contribution of carnivores jumped to 80%. This suggests that the zone of transition between a stand of emergents and open water provides a greater abundance of prey for carnivores, by giving access to both the prey associated with the vegetation and the abundant zooplankton of the open water, while maintaining an appropriate habitat.

In his review on aquatic insect food habits, Cummins (1973) concluded that "since ... most species studied seem to be nonselective generalists in their food habits, the kinds of food available and their relative proportions will essentially define the trophic relations." His conclusion stemmed from the observation that there are substantial differences in the food ingested both within genus and within a single species. For instance, the food habits of the trichopteran Glossosoma nigrior depend strongly on its habitat: in one stream it ingested mostly algae while detritus was the dominant food in another (Ibid). Age-specific variations were also found for G. nigrior: early instars consumed mostly detritus while algae dominated in later instars. As they aged, nymphs of the ephemeropteran Stenonema fuscum shifted from a diet dominated by detritus to one composed essentially of algal and animal foods (Ibid). This variation was found to be simply a function of the size of the insect; the nymphs were non-selective and fed on what they could catch and ingest.

While information about specific food preferences of aquatic invertebrates is limited to the broad categories defined earlier (algae, detritus, animal food, live vascular plant tissue), it is clear that live vascular plant tissue is rarely consumed. Macrophytes are considered as a source of food for only five families of aquatic insects (Tables 2.19 and 2.20). For example, of the 38 entries in a table of the food habits of selected aquatic insects, only 13 recorded some vascular plant tissue in the guts (Cummins 1973). In a study of the gut contents of seven species of Trichoptera, Thut (1969) found that vascular plant tissue did not constitute more than 5% of the food volume except for one species where this value exceeded 60%. The biomass of invertebrates consuming macrophytes was very small with respect to total plant standing crop in a Dutch lake, thereby suggesting that there was little utilization of the vascular plants by invertebrates (Dvorak and Best 1982). Polunin (1984) suggested that this low level of herbivory on macrophytes may be due to their poor nutritional quality as concentrations of nitrogen are limiting in these plants. While extensive plant damage by invertebrates has been recorded in some wetlands, high rates of herbivory were related to disturbances which had left the plants weakened (McDonald 1955, Brinson et al. 1981).

2.4.3.3 The detrital pathway

The lack of any significant grazing on aquatic macrophytes, combined with the high plant standing crops reached in prairie potholes, indicate that most of the plant biomass will end up as detritus (i.e. dead organic matter) in the sloughs. Most of the dominant invertebrates such as the amphipods, gastropods and many dipterans are dependent on this detritus as food. The dead plant material also provides an important structural component to the habitat of the invertebrates. The importance of nutrient recycling and decomposition processes to wetland vegetation dynamics underlines the substantial contribution of the detrital pathway to the pothole ecosystem.

The fate and transformation of dead plant tissue in freshwater systems has been reviewed by Berrie (1976) and, more recently, by Polunin (1984). The following description of the processes involved in the decomposition of macrophytes is extracted, for the most part, from these two reviews.

In prairie potholes, senescence of macrophytes begins in early July with shoot death in Carex atherodes (Davis & van der Valk 1978); the initiation of senescence in other species is staggered throughout the summer. Loss of standing litter to the water is mostly through fragmentation during the winter and spring and results from mechanical damage by wind, animals or ice-scour. Wetted litter is quickly colonized by bacteria which reach peak numbers in a few days to a month. Bacteria usually decline following this peak as they are far less abundant on highly degraded material than on fresh litter reaching water.

The number of fungal species colonizing the substrate increases with time. The increase is accompanied by a succession in mycofloral composition which may be related to changes in the detritus. While fungi and bacteria are equally important by the end of the first month of decomposition, bacteria can become the dominant microorganism in later stages as fungi may make the detritus more susceptible to bacterial colonization.

Microorganisms contribute more to decomposition than does leaching. Their role in decomposition is threefold: recycling essential nutrients, making the detritus more amenable to detritivore feeding and providing a more assimilable food in the form of their own biomass. Rapid losses of soluble forms of nitrogen and phosphorus from litter are due, for the most part, to microbial uptake. High turnover rates of phosphorus by bacteria (Currie & Kalff 1984a, b) underlines their importance in recycling nutrients within the marsh. The breakdown of structural compounds such as cellulose, hemicellulose and lignin by microorganisms make the detritus more amenable to invertebrate consumption. However, macroinvertebrates assimilate the microorganisms associated with detritus much more readily than the substrate itself. While assimilation efficiencies of detritus by macroinvertebrates are usually in the 15-30% range, those of bacteria and fungi can be greater than 90%.

The rates of microbiological decomposition of litter are affected by a number of environmental variables. While total fibre content of detritus adversely affects decay, higher nitrogen and phosphorus concentrations in the plant lead to greater use by microorganisms. The length of time during which litter is immersed in water seems to play a role in the rate of breakdown. The rate of decay increases drastically after the litter is immersed; submergents break down much more rapidly than litter which is only intermittently underwater. Also, the low light level, high acidity and low oxygen concentrations characteristic of reed beds reduce the rates of decomposition which are higher in open water.

Macroinvertebrate colonization of litter is characterized by an increase in the number of individuals and species over time. However, invertebrate changes in density probably reflect seasonal growth in populations. Various studies suggest that structural properties of the litter has much more influence on invertebrate numbers than does the fact that the substrate is decomposing.

While macroinvertebrate colonization of litter is not very different than colonization of an inert substrate, their effect on decomposition is considerable. Grazing on litter increases with time possibly because microbial colonization enhances the nutritive value of the detritus; feeding also increases with temperature. The grazing activity may reduce the number of microorganisms and alter the microbial community. However, bacterial activity (i.e. uptake rates) has been shown to be enhanced by grazing. Increases in phosphorus recycling and oxygen consumption have been noted in association with the grazing activities of a protozoan and a snail, respectively. The feeding activites of certain invertebrates can enhance microbial activity by reducing microorganism populations which are limited by nutrient availability, by releasing dissolved organic matter, by increasing microturbulence or by exposing new surfaces for colonization.

Most of the ingested detritus is not assimilated by invertebrates; the material is largely returned to the system as faeces which decompose more rapidly than the original litter. Thus, the feeding activity of detritivores leads to the breakdown of the litter into smaller particles. While the finer particles may decompose in the water column, the coarser material will end up in the sediments where further breakdown may result from consumption by benthic detritivores. Smaller particles may be utilized more readily by microorganisms.

While breakdown is enhanced by animals, most mineralization is dependent on microorganisms. Although the detritus-detritivore trophic link may not be important energetically, detritivory stimulates breakdown and microbial decomposition.

2.5 Migratory Ducks and Prairie Wetland Habitat

A substantial portion of North American waterfowl populations use the wetland habitat of the pothole region of the prairies in Canada and the northern United States during the critical reproductive phase of their life cycle. Although more than 20 species of duck are known to nest in the pothole region (Bellrose 1976), only the 11 species listed in Table 2.21 have annual breeding populations in excess of 100,000 individuals in the Canadian prairies. Most ducks using the pothole habitat are dabblers (puddle ducks). They customarily prefer semi-permanent shallow ponds and feed in the pond either at the surface or by "tipping up" so that their tails show above the water. The divers listed tend to occupy deeper more stable potholes and feed in water by diving below the water surface.

The work of wildlife ecologists conducted over the past four decades on duck populations in the pothole region has suggested both that the abundance and distribution of species is dependent on the quantity and quality of wetland habitat and that specific structural and food resource aspects of habitat quality can contribute to reproductive and recruitment success. This section will examine the intimate relations between ducks and their breeding, nesting, brood rearing and spring-summer foraging habitat in the potholes of the Canadian prairies. Our specific objectives are to elucidate the critical links between pothole ecology and waterfowl resources and to describe the indices by which reproductive success can be gauged.

2.5.1 Chronology of life history events during reproduction

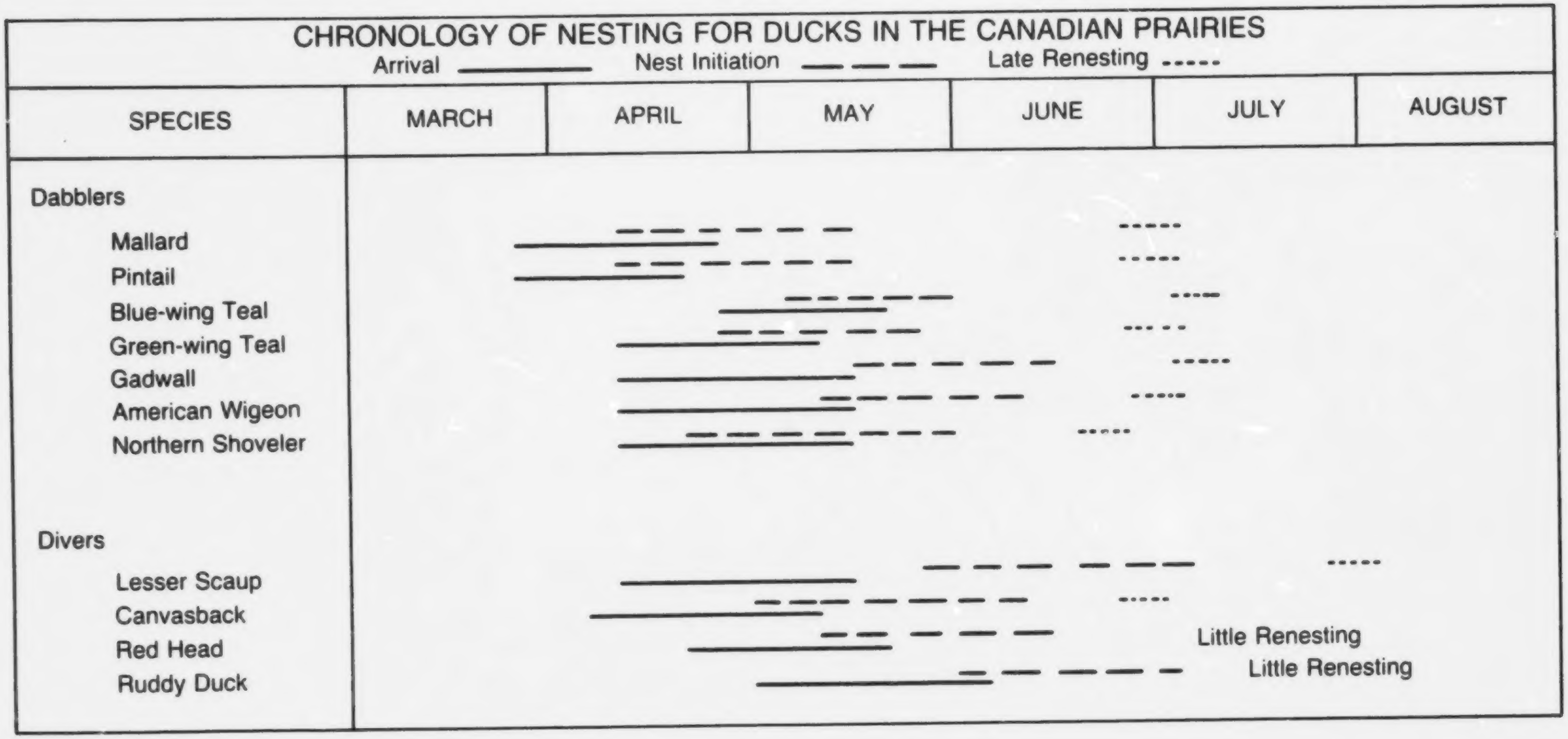
Migratory waterfowl fly north to the pothole region each spring via flyways that span the North American continent. Their arrival in the Canadian prairies is generally associated with the early melt of ice on the potholes, but the exact timing of arrival is species-specific and highly dependent on spring weather conditions as well as geographical region within the Canadian prairies (Smith & Stoudt 1968). Periods of primary arrival activity for populations of the dominant dabbler and diver species are shown in Figure 2.7. The bands on this figure represent an "average" activity period for the whole prairie, and therefore they are broader than might be expected for populations within a specific region. Mallards and Pintails are the first dabblers to arrive each spring and Canvasbacks are the first diver to mass at the potholes. These species reach the pothole region before the end of March and the majority of their breeding populations have completed the spring migrations by the end of April. Although various observers have reported Canvasbacks arriving in mass over a short period (Hochbaum 1944, Olson 1964), their arrival in the various regions may be spread from late March into early May. Large numbers of Green-winged Teal, Gadwall, American Widgeon, Northern Shoveler, Lesser Scaup and Redhead begin arriving at Canadian potholes after the second week of April and most of their in-migration is finished by mid-May. Blue-winged Teals do not reach the prairie habitat until late April. The Ruddy Duck is the last to arrive; the majority of individuals do not reach nesting areas until June.

Table 2.21. Duck species with breeding populations of greater than 100,000 in the Canadian prairies. Species within the dabbler and diver groups are listed in order of declining abundance[a].

Common Name	Scientific Nomenclature
Dabblers	
Mallard	*Anas platyrhynchos*
Pintail	*Anas acuta*
Blue-winged Teal	*Anas discors*
Northern Shoveler	*Spatula clypeata*
American Widgeon	*Mareca americana*
Gadwall	*Anas strepera*
Green-winged Teal	*Anas carolinensis*
Divers	
Lesser Scaup	*Aythya affinis*
Redhead	*Aythya americana*
Canvasback	*Aythya valisneria*
Ruddy Duck	*Oxyura jamaicensis rubida*

[a] From Cooch 1981

Figure 2.7. Chronology of arrival and nest initiation for the more common duck species in the Canadian prairies. Bars represent average activity periods for the entire prairie region.



CHRONOLOGY OF NESTING FOR DUCKS IN THE CANADIAN PRAIRIES

The arrival chronology presented in Figure 2.7 is based on normal migration conditions (average of data from Hochbaum 1944, Smith & Stoudt 1968, Bellrose 1976), however, weather conditions dictating either an early or late arrival of spring conditions may be expected to shift arrival periods accordingly. A majority of early nesting species such as the Mallard and Pintail pair during the winter and initiate nesting shortly after their spring arrival at the potholes. Nesting chronology is also summarized in Figure 2.7. As noted, Mallards are the first to nest; the early arriving flock initiates nest preparation by the second week of April prior to the complete in-migration of the bulk of the population. Similarly, the majority of Pintail initiate nest attempts between mid-April and the first week of May. Canvasbacks, the first divers to nest, follow the Pintails by approximately 1-2 weeks. Unlike the early nesting dabblers, about half of the Canvasbacks demonstrate courtship activity following the spring migration. For all the early nesters the period between arrival and nesting is short: approximately 3-5 weeks.

Blue-winged Teal populations are largely in place at the potholes by mid-May and initiate nesting activity within a month of arrival. In contrast, the Northern Shoveler arrives in early April but does not initiate nesting until the latter half of May. Redhead populations follow the arrival-nesting chronology of the Shoveler. The Redhead, however, is unique in that it may drop eggs into the nests of other species (Hochbaum 1944) and is therefore subject to their temporal nesting habits. Gadwalls begin nesting in the latter third of May and continue to the middle of June. The Lesser Scaup and Ruddy Duck are late nesters: nest initiation does not begin in earnest until the latter half of June and may continue into the first week of July. These late nesters continue to select mates on the nesting grounds and may delay initiation of their nesting activity for 30-40 days after arrival.

Most species renest if early attempts fail (Stoudt 1971, Smith 1971). Using marked Canvasback hens, Doty et al. (1984) observed nearly 60% renesting within a 5 to 30-day interval following nest failure. The clutch produced in renesting attempts is generally smaller than that produced by successful first nesters. Renesting for most dabblers and Canvasbacks may carry on into early July in most years. Other divers such as the Lesser Scaup and Redhead can renest up to late July. Ruddy Ducks probably do not attempt renesting in most years. The chronology of renesting described here (see Figure 2.7) indicates that most prairie species have adopted an extended nesting strategy to deal with the adverse conditions which occur with long periods of either wet weather or drought. Under extreme conditions, such as the drought of 1949, ducks deserted nests and made no attempts to renest (Smith & Stoudt 1968). It should be noted that early nesters may hatch clutches prior to mid-season drought but are potentially subject to spring drought or blizzards. On the other hand, late nesters are certainly at risk of drought and early freezes.

Incubation periods for the major divers and dabblers range from 20-30 days (Hochbaum 1944). The total two-month nesting period places female ducks and embryos at great risk. The female and incubating eggs are away from the comparative security of the water and therefore more subject to predator threat. In addition, they are at the mercy of adverse climatic events and upland nests may be exposed to damage from cultivation.

Mallard and Pintail eggs which survive natural and mechanized destruction hatch between the last week in May and mid-June (Figure 2.8). Blue-winged Teal, Northern Shoveler and Canvasback broods hatch during the latter half of June. Gadwall hatch is from the last week in June through mid-July. Lesser Scaup and Ruddy Duck broods hatch between the first to last week in July. The Redhead hatch is intermediate between the early and late nesting divers. Hatch of successful renesting efforts occurs primarily during June and July for early nesting species and into August for the late nesters.

Upon hatch, broods are flightless and will be for the next 40-75 days (see Hochbaum 1944, page 108, for species-specific data on age to attain flight). During the flightless period of rearing, broods are highly dependent on their immediate wetland habitat for protective cover and food resources. The majority of Pintail, Mallard and Teal broods fledge from the latter half of July into the initial week of August (Figure 2.8). Shoveler broods fledge in mid-August followed in late August through mid-September by the peak fledge of Gadwall, Lesser Scaup, Canvasback and Redhead. All dabbler broods should have attained flight by the end of September. Broods of diving species may still be flightless in October; those broods which are late to fledge are at risk of being caught in an early freeze.

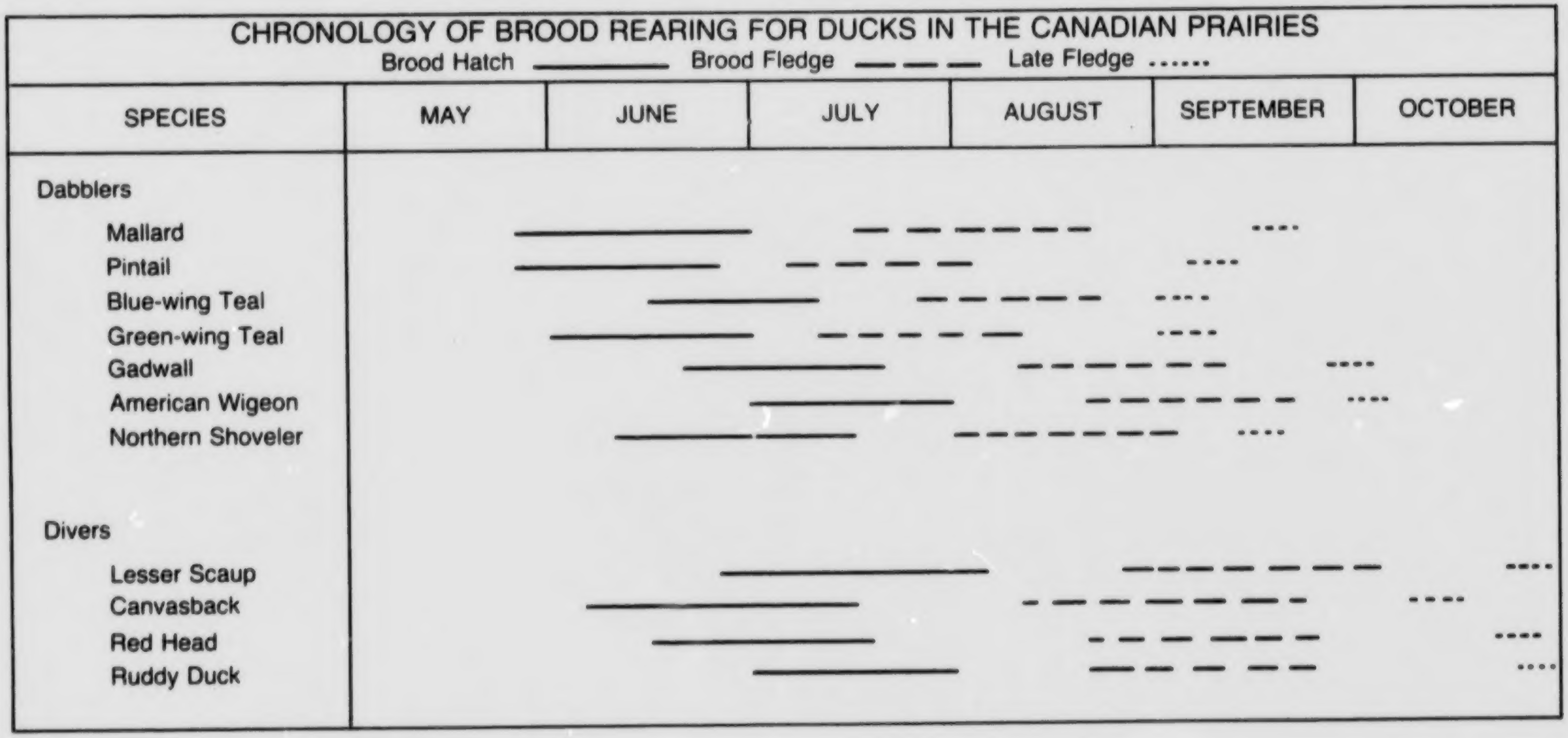
2.5.2 Habitat requirements

General habitat quality for ducks is particularly crucial during the period of nesting and brood rearing. Specific features of pothole size, water and sediment composition, types of margin and surrounding land use as well as floral and faunal composition affect nest and brood success. Because resource requirements at crucial links in the reproductive and recruitment process are specific and because the necessary food and nesting habitat resources of the potholes may vary substantially for both natural and cultural reasons, a precarious ecological balance exists for the large numbers of waterfowl using prairie pothole habitats.

2.5.2.1 General needs

Upon spring arrival at the potholes, ducks are in need of open water habitat for foraging, and slough and upland vegetation for nesting and protective cover. Duck requirements for shallow water bodies during reproduction is universal. On large wetland areas, such as Delta Marsh,

Figure 2.8. Chronology of brood hatch and fledge for the more common duck species in the Canadian prairies. Bars represent average activity patterns for the entire prairie region.



CHRONOLOGY OF BROOD REARING FOR DUCKS IN THE CANADIAN PRAIRIES

Manitoba, pair densities of most species were correlated with the proportion of shallow marsh habitat (Kaminski & Prince 1984). A number of reports correlate population declines with drought and reduced numbers of potholes (Smith 1971, Stoudt 1971, Boyd 1981, Leitch & Kaminski 1985). Smith (1971) states that "no single cause for reducing waterfowl production in the Saskatchewan parklands was so important, so far-reaching in its effects, so destructive of habitat or so detrimental to the well-being of ducks as drought." Lynch (1984) stated, in clear but colorful terms, that duck production can only be high ("super crop") in the few wet years when the numbers of aquatic habitats in the southern prairie is greatly expanded; duck production is substantially reduced ("no crop a-tall") if drought dries up potholes in the parklands. According to Lynch, the more northern parklands tend to have a more reliable but "mediocre" production except in severe drought conditions, whereas the southern prairie is a very unreliable habitat for duck production due to its vulnerability to climatic fluctuations. It should be remembered, however, that drawdowns lead to the revitalization and enhanced production of wetlands on a long-term basis. Lynch attributed the mediocre production in the parklands to heavy avian predation on duck nests.

Dabblers and divers form two distinct ecological groups in their use of potholes. From 40-54% of populations of dabbling species, but only 9-19% of diving species, occupied seasonal ponds in a Stewart and Kantrud (1974) study. Conversely, semi-permanent and permanent ponds provided breeding habitat for 75% of diving ducks but less than 30% of dabbling ducks. Stewart and Kantrud (op. cit.) reported that >75% of the total breeding ducks in the North Dakota pothole region were found on seasonal and semi-permanent ponds and lakes with the remainder on impoundments, streams and other habitats such as ditches. In a subsequent survey, more than 80% of duck broods in this region were also found on seasonal and semi-permanent wetlands (Duebbert & Frank 1984). Use patterns appear to be much the same in the Canadian parkland potholes (Smith & Stoudt 1968). Table 2.22 summarizes pair densities by species on the various wetland types. Only Blue-winged Teals and Pintails made substantial use of ephemeral ponds and only Blue-winged Teals utilized all types to a moderate extent. Diving ducks did not inhabit ephemeral or temporary wetlands.

Some species-specific habitat preferences are summarized in Table 2.23. The Mallard is somewhat of a generalist in terms of its use of pothole types. Breeding pairs and broods may forage in shallow depressions as well as in deeper more stable seasonal, semi-permanent and even permanent ponds. Dwyer et al. (1979) reported that laying hens spent 45% of their time foraging while on seasonal wetlands. In the Canadian prairie parklands, potholes of <0.5 ha were preferred by breeding Mallards and broods. Cooch (1969) suggested that Mallards had good success in potholes <0.25 ha. The Mallard appears to be a species which readily prefers aquatic habitat surrounded by cultivated cropland (Smith 1971). Flooded whitetop, sedge and bulrush beds provide the favored cover for Mallard broods, however, they are the most tolerant of open water and bare shores of all species except the Pintail (Bellrose 1976).

Table 2.22. Density (pairs/km^2 of wetland) of breeding ducks on wetlands containing ponded water[a].

SPECIES	Wetland Type				
	Ephemeral[b]	Temporary[b]	Seasonal[b]	Semi-permanent[b]	Permanent[c]
Dabbling Ducks					
Mallard		66.7	44.9	28.6	4.3
Gadwall		54.3	38.8	29.5	7.9
Pintail	49.2	82.4	58.6	27.0	3.6
Green-winged Teal		6.0	13.1	5.2	1.7
Blue-winged Teal	137.7	112.6	122.4	73.7	14.5
Shoveler		36.2	34.4	22.7	1.7
American Widgeon		6.0	5.0	2.4	2.0
Diving Ducks					
Redhead			7.4	29.6	3.3
Canvasback			1.6	8.1	2.2
Lesser Scaup			2.9	4.7	6.1
Ruddy Duck			4.0	28.2	4.8
Total dabbling ducks	186.9	364.2	317.2	189.1	35.7
Total diving ducks			16.6	73.2	17.2
Area (km^2) of habitat during breeding					
Min.	0.0007	0.449	9.66	7.70	6.91
Max.	0.0203	0.509	10.65	9.03	6.99

[a] Modified from Kantrud and Stewart 1977

[b] Data from random plots studied 1967-69

[c] Data from 1965 on subjectively selected wetlands added to 1967-69 results

Table 2.23. Habitat preferences of breeding ducks in the prairie pothole region.

	Pothole Characteristics				
Species	Size Class (ha)	Permanence	Vegetation	Surrounding Land Use	References
Dabblers					
Mallard	0.5	seasonal to semipermanent ponds preferred, however the Mallard is a generalist and may also use both more transient and larger more stable aquatic habitats	some emergent aquatic vegetation required; ponds partially to completely surrounded by woody vege-	cropland	Smith 1971 Stoudt 1971 Stewart & Kantrud 1973 1979 Talent et al. 1982
Pintail	rather evenly distributed with size class from 0.5 to 20	seasonal to semipermanent ponds preferred; stock watering ponds used particularly in dry years	pond with more open shoreline preferred; no discernible preference for or against woody surrounding vegetation	grazed grassland or woodland	Stoudt 1971 Stewart & Kantrud 1973
Blue-winged Teal	0.5	seasonal to semipermanent ponds and dugouts with characteristics of semipermanent ponds preferred	pond with some emergent aqautic vegetation selected; wooded ponds used nearly as frequently as open ones	ungrazed grassland and woodland	Smith 1971 Stoudt 1971 Stewart & Kantrud 1973
Gadwall	1.0 to 2.0	seasonal and semipermanent ponds preferred	ponds with emergent aquatic vegetation at their fringe and with dense upland vegetation for nesting preferred; potholes with large amounts of edge per unit area of pond selected	ungrazed or mowed vegetation in grasslands	Stewart & Kantrud 1973 Bellrose 1976 Hines & Mitchell 1983 Duebbert & Frank 1984

Table 2.23 (cont'd)

Species	Pothole Characteristics				References
	Size Class (ha)	Permanence	Vegetation	Surrounding Land Use	
American Widgeon	1.0 to 2.0	primarily seasonal and occasionally semipermanent ponds utilized	some emergent aquatic plants required; dense peripheral woody vegetation preferred	ungrazed grassland	Smith 1971 Stewart & Kantrud 1973
Northern Shoveler	0.5 to 2.0	seasonal to permanent ponds preferred	ponds with partial to well-developed emergent vegetation selected; open grassland surroundings preferred	grazed grassland or cropland	Stewart & Kantrud 1973 Poston 1974 Duebbert & Frank 1984
Divers					
Lesser Scaup	1.0	semipermanent to permanent ponds preferred	ponds with well developed emergent aquatic vegetation preferred; pond partially enclosed by trees and shrubs widely used	ungrazed grassland	Smith 1971 Stewart & Kantrud 1973
Canvasback	0.5 to 2.0	semipermanent to permanent ponds and dugouts preferred	ponds with well developed emergent vegetation but little surrounding woody vegetation are selected	equal use of grazed grasslands, woodlands, hay fields	Stoudt 1971, 1982 Sugden 1978
Redhead	2.0	semipermanent to permanent ponds preferred	ponds with lush aquatic vegetation on edges selected		Stewart & Kantrud 1973 Bellrose 1976

Table 2.23 (cont'd)

	Pothole Characteristics				
Species	Size Class (ha)	Permanence	Vegetation	Surrounding Land Use	References
Ruddy Duck	2.0	semipermanent to permanent ponds and marshes preferred	ponds with well developed emergent vegetation selected		Stewart & Kantrud 1973 Bellrose 1976

Pintails showed little obvious size preference for potholes but tended to occupy ponds with open shoreline (Stoudt 1971). Blue-winged Teals generally utilized seasonal and semi-permanent potholes which were ringed with emergent macrophytes and often <0.5 ha in size and as deep as 1 m (Stewart and Kantrud 1973). The American Widgeon and Gadwall showed similar preferences for size, permanency and development of emergent vegetation and both species showed some preference for ponds densely bounded by woody vegetation (Smith 1971, Bellrose 1976). The Northern Shoveler used potholes of a broad range of sizes but prefered ponds of a more permanent nature; they require open water for their unique plankton feeding habits. This species tolerates grazing and cultivation activity in proximity to its core habitat.

The diving species prefer larger and deeper, thus more stable, pond habitats than those widely utilized by dabbling species (Table 2.23). Lesser Scaups, Canvasbacks, Redheads and Ruddy Ducks prefer semi-permanent and permanent ponds. Lesser Scaups at Lousana, Alberta, preferred potholes of approximately 3 m depth and 1 to 2 ha in size, partially surrounded by trees and located in ungrazed grasslands (Smith 1971). Canvasbacks utilize a range of sizes and types of potholes: larger, more permanent ponds for feeding, resting and courting and smaller, shallower, seasonal ponds for nesting (Trauger & Stoudt 1978). Brood ponds may be intermediate in size between the foraging and nesting ponds and usually are ringed by well-developed emergent marsh vegetation. Redheads and Ruddy Ducks use large potholes and marshes and require well-developed emergent vegetation for nesting (Bellrose 1976).

Although some generalities may be made in regard to wetland preferences by duck species, recent observations and radio-tracking of breeding pairs and broods of dabblers indicate that a complex of natural wetlands with a sizable seasonal component may be essential to their reproductive and recruitment success (Dwyer et al. 1979, Talent et al. 1982). Hens, drakes and broods may use a number of potholes in an area. Favorable habitats within the complex contain a balance of protective cover and appropriate nesting vegetation as well as aquatic invertebrate food resources to supply adequate nutrition to laying hens and young ducklings.

2.5.2.2 Nesting habitat

The physiognomy of vegetation surrounding a pothole or marsh is an important factor contributing to its acceptability as a nesting, foraging or brood rearing site for ducks. Waterfowl biologists have suggested that undisturbed abundant nesting cover associated with certain potholes is the factor responsible for the greater numbers of nesting ducks at these sites (Keith 1961). Griffith (1948) concluded that lack of cover is more often the factor limiting reproductive success than lack of food. Nest success is primarily a function of predation, and cover is clearly important to protect ducks from predators during the period of reproduction, a time at which they are particularly vulnerable. Predation is the major source of egg mortality and nest loss (Cowardin & Johnson 1979). Cowardin et al. (1985) estimated that predation accounted for nearly 70% of the unsuccessful Mallard nests monitored in North Dakota. Recent unpublished evidence suggests that predation may account for nearly 90% of nest loss in many areas of the Canadian prairies (R. Greenwood, U.S.F.W.S. Northern Prairie Wildlife Research Center, pers. comm.). It should be noted that agricultural cultivation has tended to confine predators and ducks together in small areas of suitable habitat.

A number of studies have clearly demonstrated higher habitat occupancy, particularly by dabbling ducks, in potholes with undisturbed upland vegetation. Parkland potholes completely surrounded by woody vegetation supported twice as many dabbling pairs as did potholes in agricultural areas where adjacent upland vegetation was disturbed (Dwyer 1970). In intensively farmed regions, the greatest diversity of nesting species occurred in untilled uplands (Higgins 1977). The mowing of upland grasses reduced the number of nesting dabblers by half in a controlled study in the North Dakota pothole region (Martz 1967). Likewise, heavy grazing of uplands restricts their acceptability as nesting habitat for dabbling ducks (Kirsch 1969). The study by Dwernychuk and Boag (1973) established a link between the use of herbicides on upland vegetation and consequent reduced nesting activity of ducks. Applications of a 2,4-D ester at < 2 kg ha^{-1} to transects of upland vegetation on two islands in Miquelon Lake, Alberta, reduced the number of broadleaf plants and led to a substantial decline in the number of nesting ducks. The critical importance of well-developed vegetative cover for the nesting success of ducks is shown quite clearly in data and conclusions from a two-factor study by Duebbert and Kantrud (1974): idle (undisturbed) grassland containing tall, dense rank cover, without predator reduction, produced six times as many ducks as areas with upland disturbance and substantial predator reduction. They concluded that the establishment and maintenance of excellent upland cover was a sounder management practice, contributing to greater duck production, than the direct reduction of predators.

Our focus on the importance of upland vegetative cover to the nesting success of ducks reflects the dominant nesting strategy in prairie duck populations. All of the numerically dominant dabbler species are upland nesters as is the Lesser Scaup, the most abundant of the diving ducks in the Canadian pothole region. This initial emphasis on upland nesting cover does not discount the importance of emergent macrophytes and woody vegetation as over-water nesting habitat for other diving duck species and as protective cover for feeding broods.

A summary of the habitat requirements for nesting dabblers and divers is presented in Table 2.24. Mallards are primarily upland nesters, although more generalized in their selection of appropriate nesting sites than any other dabbling duck species. Their primary requirement appears to be dense vegetation about 60 cm high, regardless of plant composition (Bellrose 1976). Mallards nested in all types of vegetation at Redvers, Saskatchewan, but preferred grassland vegetation (Stoudt 1971). Wheat and quack grass growing in areas too moist to be plowed in the spring were frequently used for nest cover. New growth of woody vegetation, such as snowberry clumps, provided an alternative nesting habitat for Mallards in the Redvers parklands. When Mallards built over-water nests, cattails and bulrushes provided nest support and cover (Krapu _et al_. 1979). All over-water nests were located in the shallow periphery of potholes (Stoudt 1971). In the Alberta parklands, grazed and ungrazed buckbrush clumps were the most highly selected nesting habitat. Open grasslands and willow or aspen bluffs also rated high as Mallard nesting habitat: these types of cover contained approximately 75% of Mallard nests (Smith 1971). In southeastern Alberta, rushes such as _Juncus_ provided the principle nesting cover (Keith 1961); whitetop river grass was preferred at Delta Marsh, Manitoba (Bellrose 1976). In agricultural areas, hay fields are preferred by nesting Mallards (Duebbert 1969). A high density of vegetation cover to screen nesting Mallard hens appears to be particularly critical in areas such as islands where the potential for avian predation or conspecific harassment is great (Lokemoen _et al_. 1984).

Pintails are somewhat unique among ducks in their preference for upland nesting habitat where vegetation is low and sparse. Also, they locate their nests farther from water than other upland nesting species (Bellrose 1976). Pintails at Redvers had a tendency to seek open areas for nest building (Stoudt 1971). Vegetative cover for 70% of the nests was composed of grasses, snowberry clumps and cattail-sedge assemblages. In intensively farmed areas, Pintails, unlike other dabblers, frequently used standing stubble as nesting cover (Higgins 1977). At Delta Marsh, Pintail nests were abundant in stands of whitetop river grass and blue grass (Sowls 1955). Near Brook, Alberta, nests were frequently found in rush beds, mixed prairie grasses and burned weed areas (Keith 1961). More than any other waterfowl species, Pintails accept farmed land as suitable nesting habitat (Bellrose 1976).

Blue-winged and Green-winged Teals display more selectivity than Mallards or Pintails in choosing a nesting habitat. Nearly 80% of Blue-winged Teal nests at Redevers were located in grasslands (Stoudt 1971). Tall, dense stands of wheat, brome and quack grasses provided necessary cover. Shorter bluegrasses were favored in heavily grazed pastures. Grass, sedge and buckbrush provided preferred nesting cover at the Lousana area in the Alberta parklands (Smith 1971). Blue-winged Teal nests were nearly equally divided between untilled upland and growing grain fields in a study confined to an intensively cultivated area (Higgins 1977).

Table 2.24. Nesting habitat requirements summarized for duck species in the Canadian prairie regions[a].

Species	Nest Site Description	Comments
Dabblers		
Mallard	prefer upland to marsh sites; nest location may vary from 1 to 100 m from pond edge; tend to select cover about 60 cm high, including grasses (particularly quack grass), buckbrush, snowberry clumps, whitetop river grass, sweet clover, willow and aspen; will nest on floating cattail beds near pothole edges	generalist in site selection
Pintail	select open upland areas for nesting where vegetation is low or sparse; bare ground in stubble fields is also used for nesting; nests are located far from water, usually more than 100 m and sometimes more than 1 km; vegetative cover is variable and includes whitetop river grass and blue grasses, buckbrush, mixed prairie grasses or weeds, rushes, hay crops and grain stubble	nests widely in farmland habitats
Blue-winged Teal	favor grasses for nest sites; nest location may vary from 20-200 m from water; preferred cover includes hay fields, mixed grass (wheat, brome, quack grass), buckbrush; sedges are locally important	highly selective of grasses
Green-winged Teal	prefer upland nest sites located at distances of 1-100 m from water; use available cover but favor dense stands of grass, weeds or brush	

Table 2.24 (cont'd)

Species	Nest Site Description	Comments
Gadwall	prefer upland sites for nesting generally within 100 m of water; choice of cover depends on what is available but a certain height and thickness of vegetation is required; selected plants include whitetop rushes and buckbrush; nettles and thistles used locally	seek tall dense vegetation for nesting
American Widgeon	usually select an upland site located from a few meters to 400 m from water; preferred cover includes buckbrush and rushes	
Northern Shoveler	prefer upland sites located within 20-60 m of the nearest pond; grasses are selected as vegetative cover and shortgrasses are preferred to tall species; nests also located in alfalfa fields or rush beds	weedy patches and woody vegetation are avoided as are grain fields
Divers		
Lesser Scaup	more prone than other diving species to select an upland nesting site; most nests are within 20 m of water; sedges are preferred vegetative cover, rushes are also used as are mixed prairie grasses; forbs as well as willow and aspen clumps are used to a lesser extent; Scaup avoid small ponds completely surrounded by trees which interfere with their low circling flight behaviour prior to landing on water	taller vegetation (30 to 60 cm) selected for concealment in areas with large numbers of avian predators

Table 2.24 (cont'd)

Species	Nest Site Description	Comments
Canvasback	select nest sites in a diversity of aquatic habitats including large marshes, ponds, potholes; prefer shallow less-enduring ponds (15-60 cm deep) for nesting; over-water nest cover preferences include cattails, bulrushes, whitetop river grass, willows and sedges	willows important as other cover is flooded during wet years
Redhead	select nest sites in emergent vegetation of large marshes or larger, deeper potholes; nests are over water in dense stands of plants; preferred cover is hardstem bulrush with cattails and sedges selected less frequently; Redheads appear to have specific plant requirements not merely satisfied by a growth form that provides adequate concealment	site requirements appear stringent
Ruddy Duck	select nest sites in both large and small marshes; nests are over water in emergent vegetation found at approximately 30 cm water depth; cover is often provided by sedges, *Juncus* and bulrushes characteristic of particular water body	select cover growing at particular water depth

[a] Descriptions based primarily on information summarized in Bellrose 1976, Smith 1971 and Stoudt 1971

Gadwalls seek tall, dense vegetation for nesting more than do other dabbling ducks. Hines and Mitchell (1983) reported that nesting pairs were most numerous in potholes with well developed edge vegetation. The preferred cover was whitetop river grass at Delta Marsh (Sowls 1955) and rushes like Juncus at Brooks (Keith 1961). The use of upland grasses, nettles and other weeds as nesting cover by Gadwalls has also been reported (Gates 1962). On Millers Lake Island, North Dakota, thick shrubs such as snowberry and woodrose were selected as shelter (Duebbert et al. 1983).

American Widgeons are also upland nesters. Their nests were most often located in Juncus beds at Brooks (Keith 1961). Other selected plant cover included woody clumps of buckbrush or greasewood (Bellrose 1976).

Various types of grasses make up the favored vegetative cover for Northern Shoveler nests, but data indicate a preference for short growth forms over taller species (Girard 1939). Pasture grasses were most frequently used for nesting in the Strathmore study area, Alberta, however, nests were also located in alfalfa fields and rush beds surrounding potholes in this area (Poston 1974). Shovelers avoided nesting in weedy patches or in clumps of woody vegetation (Bellrose 1976).

Lesser Scaups are more prone to nest on upland habitat than are other diving ducks. Scaup selected sedges over other vegetation for nest cover at Lousana (Smith 1971) and in a study in the Saskatchewan Delta (Townsend 1966). They nested extensively on floating mats of vegetation in the Delta Marsh (Bellrose 1976). Stands of rushes and seasonally mixed prairie grasses were selected cover types for nesting Scaups near Brooks (Keith 1961). Lesser Scaups tended to avoid nesting in forest-rimmed potholes in the Manitoba parklands (Dwyer 1970). Tall trees appeared to interfere with their pattern of low circling flight prior to landing on potholes.

Canvasbacks nest over water in a diversity of habitats. At Redvers, 80% of Canvasback nests were located in cattails; sedges and woody plants were of secondary importance as nest cover (Stoudt 1971). Willows were used but primarily during wet years when other cover was flooded. A similar selection of emergent vegetation was reported for Canvasback populations at Lousana (Smith 1971). Sugden (1978) also cited the importance of cattails and whitetop river grass habitat for nesting, but reported that Canvasbacks also nested successfully in heavily wooded ponds with sparse nesting material. Their success in this less preferred habitat may have been due to reduced nest parasitism by Redheads. At Delta Marsh, bulrush beds were favored nesting cover (Hochbaum 1944) and on the Saskatchewan Delta, most nests were found in phragmites cane (Townsend 1966).

Redheads are over-water nesters which locate sites in larger and deeper potholes. They display a strong preference for hardstem bulrush beds over other types of vegetation; cattails and sedges are less frequently used as support and cover. Bellrose (1976) suggested that the Redheads' requirement of bulrush habitat was specific and stringent, and not merely based on a vegetative growth form providing adequate concealment.

Ruddy Ducks select both large and small potholes or marshes for nesting and tend to nest on emergent vegetation characteristic of the particular aquatic system (Bellrose 1976). Plant cover growing at the most desirable water depth (25-30 cm) was more important in the selection of a nest site than the qualities of the plant cover (Low 1941).

As a group, upland nesters, particularly those requiring tall dense cover, are vulnerable to physical or chemical agricultural practices which remove or reduce vegetation at slough margins and on uplands in close proximity to pothole habitat. Not all upland nesting species nest preferentially in the emergent vegetation and moisture-loving sedges immediately surrounding slough margins. Several species (e.g. Pintails) nest in upland vegetation sometimes several hundred yards away from any wetland and broods are taken to the wetlands after hatching.

In days prior to extensive agricultural development, the Canadian prairie upland habitat was characterized by native grasses, buckbrush, silverberry, raspberry, gooseberry and aspen (Smith et al. 1964), but upland nesting sites are now likely to be intimately associated with cropland. Milonski (1958) stated that most studies of nesting waterfowl were conducted on marshes and the immediate surroundings and therefore little was known of the importance of farmland for nesting. This and other research have since documented the use of farmland by nesting waterfowl, but the information is still patchy and often contradictory. However, the following general principles emerge. Apart from the slough margin, the only adequate nesting cover on farmland early in the nesting season is stubble from the previous year's growth. However, birds which establish their nests in stubble are highly vulnerable to any spring agricultural activities such as tilling, planting or disking of the land left fallow. For example, Martz (1967) estimated that 70% of nest losses in the pastures and fallow fields studied were due to farming activities. For this reason, it has been postulated that a reduction in summer fallow would benefit waterfowl nesting (Zittlau 1979).

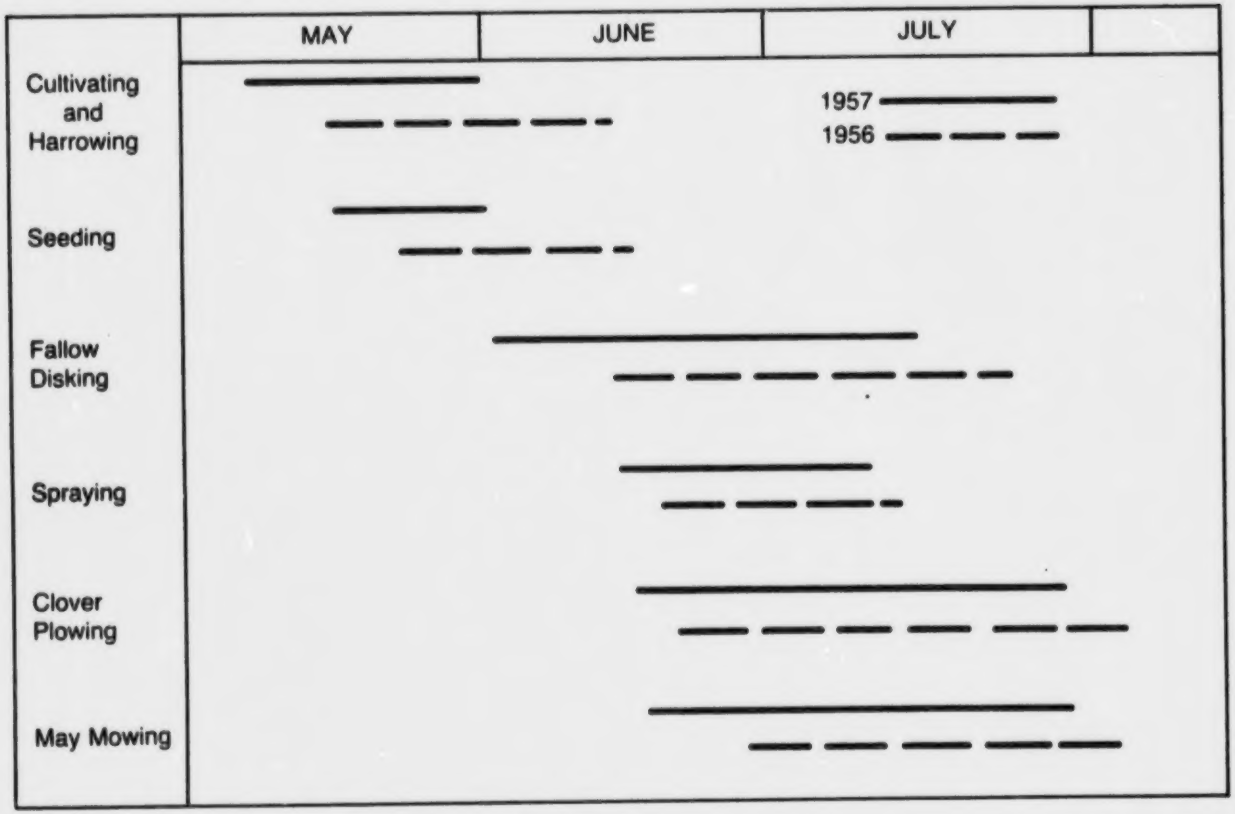
It is clear that the likelihood of successful breeding on farmland is highly dependent on the crop type and exact cropping practices (Table 2.25). Not surprisingly, the highest nest densities and nesting success tend to be found in crop types with the best cover, the most ground litter and the least amount of disturbance. Grain crops would probably be adequate for nesting were it not for the poor timing of the cultivation operations relative to the nesting phenology. The time of cultivation practices in the Portage Plain in Manitoba is shown in Figure 2.9. Some overlap between the timing of cultivation practices and the nesting

Table 2.25. Summary of key studies relating nest density and nest success of dabbling ducks to crop type.

Crop type		Nest density/ha or % use of crop type relative to total population	Nesting success	Reference
Wheat crop		5.8%	0%	Evans and Wolfe 1967
Wheat stubble		5.8%	0%	
Pasture and hay		20.0%	25%	
Alfalfa		33.0%	21%	
Roadsides		28.0%	46%	
Alfalfa and fallow[a]		.68/ha	8%	Martz 1967
Meadow[a]		.68/ha	25%	
Roadsides and shorelines[a]		1.4/ha	43%	
Native grassland (grazed)		.07/ha	14%	Kirsh 1969
Native grassland (ungrazed)		.11/ha	28%	
Grain stubble		.05/ha	0%	Duebert and Kantrud 1974
Hay		.37/ha	50%	
Pasture		.14/ha	57%	
Tame hay		.25/ha	64%	
Undisturbed grass-legume cover		.84/ha	68%	
Summer fallow	annually tilled cropland	.01/ha	8%	Higgins 1977
Mulched stubble		.02/ha	9%	
Standing stubble		.04/ha	15%	
Growing grain		.01/ha	40%	
Idle grassland	untilled upland	.16/ha	25%	
Grazed pasture		.04/ha	0%	
Idle pasture		.15/ha	50%	
Roadsides		.36/ha	19%	
Idle alfalfa		.17/ha	22%	
Dry wetland basins		.14/ha	44%	

[a] Portions of each crop type mowed before the breeding season

Figure 2.9. Chronology of agricultural activities in the Canadian prairies (from Milonski 1959)


MAY
JUNE
JULY
Cultivating and Harrowing
1957
1956
Seeding
Fallow Disking
Spraying
Clover Plowing
May Mowing

periods for ducks (see Figure 2.7) is expected for all but the earliest nesters. Mallard, Pintail, Shoveler and Teal nesting may be disturbed by cultivating and seeding activities over much of the prairie region.

Agricultural practices which reduce tillage and are therefore less disruptive to vegetative cover may contribute to improved nest success for ducks. Cowan (1982) reported that duck nest densities were higher on zero-tillage farms than conventional farms: success rates were 42% for nests on non-tilled land and 13% for nests on cultivated farms. The data indicate that improved cover associated with no-tillage agriculture can provide both more and better habitat for farm-nesting ducks. Recent productions estimates by Sugden and Beyersberger (1985) suggest a clear advantage of zero-tilled fields over conventional stubble fields for nesting ducks.

2.5.2.3 Brood vulnerability to predators

Duckling mortality can be substantial: in a study at Kindersley, Saskatchewan, nearly 70% of successfully hatched clutches of Mallards were lost prior to fledging (Dzubin & Gollop 1972). In contrast, Cowardin and Johnson (1979) estimated that an average of 70% of Mallard broods survived from nest to water. Because there is not sufficient field data available, it is not possible to determine the relative importance of various sources of brood mortality such as weather factors, predation or food abundance. Rearing habitat preferences often reflect both feeding and predator avoidance strategies. This section will deal specifically with those factors which influence the vulnerability of dabbler and diver broods to predation losses prior to fledging. Loss of entire broods on water can occur as the result of mink predation (Talent et al. 1983).

Dabbler broods generally require emergent vegetation for cover near foraging sites (Table 2.26). Hines and Mitchell (1983) reported that Mallard, Pintail, Blue-winged Teal and Shoveler broods were generally within 23 m of shoreline cover while feeding, whereas broods of Gadwall and diving species often fed at 60-75 m from the closest cover. Gadwall broods, in contrast to diver broods, avoided open water areas during midday. Of the dabblers, Mallard and Pintail broods are the most tolerant of open water and bare shorelines. The habit of divers to submerge to avoid predators is reflected in their obvious preference for deep open water habitats.

Beyond species preferences for vegetative cover or open water, little specific information on brood habitat requirements is available. It is assumed that the basis of selection of individual ponds for brood rearing is food resource availability and proximity to the nest site, particularly during the ducklings' downy phase.

The mobility of broods or, more specifically, the amount of time spent traveling overland, is directly related to the risk of predation mortality. A negative correlation has been found between the distance

Table 2.26. Descriptions of brood habitat and brood mobility[a].

Species	Brood-rearing Habitat Description	Mobility
Dabblers		
Mallard	Flooded white top river grass, sedge and hardstem bulrush beds are favorite cover for broods; but this species is more tolerant of open water and bare shores than other dabbling ducks except Pintail; pond size preference varies regionally but consistently includes ponds with reasonably stable water level and well developed emergent vegetation	Tend to use a number of potholes; broods remain no longer than 20 days on a pothole; may move as far as 0.8 km/day
Pintail	Ponds from 0.5-1 hectare in size or larger and of sufficient depth to maintain water throughout the brood growth period preferred; well developed emergent vegetation for protective cover sought, although broods are very accepting of bare shoreline; stock watering ponds often provide acceptable rearing habitat	Hens are highly mobile and move broods farther overland than other prairie ducks; brood remains no longer than 14 days on a pothole
Blue-winged Teal	Prefer semipermanent potholes often >1 m deep and ringed with emergent vegetation; ponds often located in grasslands or hay fields	Brood travels overland from the nest to reach foraging sites but often remains ensconced once there
Gadwall	Heterogeneous areas with tall emergents and open water favored; broods will feed at greater distances from cover than broods of other dabbler species	Favored feeding areas are at times substantial distances from the nest site, necessitating long overland treks (0.5-2.0 km), although this would not appear to be the norm in prairie potholes

Table 2.26 (cont'd)

Species	Brood-rearing Habitat Description	Mobility
American Widgeon	Prefer larger potholes ($\ll$1 ha) with relatively stable water level and well developed emergent vegetation	Broods thought to be more sedentary than those of other dabbler species
Northern Shoveler	Rearing areas vary but shallow ponds containing submergent and emergent vegetation preferred	Broods highly mobile and seldom remain on any pond for more than 7 to 10 days
Divers		
Lesser Scaup	Favor larger, more permanent potholes ($\ll$1 ha); ponds with partially wooded shorelines for wind protection preferred; emergent vegetation not essential for brood cover	Broods not highly mobile
Canvasback	Broods seek open water of the largest and deepest potholes; wetlands in pastures receive heavy use	Among the most mobile of duck broods; seldom spend more than 7 days on a pothole
Redhead	Broods seek open water areas	Broods moderately mobile
Ruddy Duck	Approximately one third of broods feed in deep marshes, the remainder in open areas	Broods highly sedentary and infrequently travel overland

[a] Data taken primarily from Bellrose 1976, Smith 1971, Stoudt 1971

moved overland by young broods and the losses to predation (Ball et al. 1975). When ducks move frequently and over great distances between nesting and brooding habitats, losses of entire broods are frequent (Dzubin & Gollop 1972, Reed 1975). Pintails, Mallards and Shovelers lose a high number of whole broods to predators because of their nomadic behaviour (see Table 2.26). Blue-winged Teals, Gadwalls and Widgeons are more sedentary in habit than other dabblers and are expected to sustain lower losses to terrestrial predators. The Ruddy Duck is the most sedentary of the major diving species while the Canvasback is the most mobile (Bellrose 1976).

The relative importance of cover to brood success may have increased in recent years with destruction of habitat. Mallard broods, because of their catholic distribution, early hatch and high mobility, are currently highly vulnerable to predation.

2.5.3 Food resources during reproduction

The importance of the quantity and quality of food resources in the pothole habitat during the nesting and brood-rearing period to the reproductive success of ducks is being increasingly documented. During the reproductive period, adults and ducklings are highly dependent on food, particularly aquatic macroinvertebrates, from the aquatic habitat. The food habits of a reproductively active female and newly hatched duckling are, in many cases, substantially different from those of other adults and immatures. For example, dabblers such as Pintails feed primarily on plant food for most of the year but select animal food during the period of egg formation (Krapu 1974a, b) and early duckling growth (Sugden 1973). Feeding ecology studies on a number of dabbling and diving ducks indicate that these periods of high dependency on aquatic invertebrate prey are a normal part of the reproductive cycle of all duck species and are largely governed by the nutritional requirements of each stage.

2.5.3.1 Laying female

During the period of egg formation, food intake by female ducks must be sufficient to supply normal metabolic needs and to provide the nutrients necessary to produce viable clutches of eggs. Laying Mallards, Pintails and Gadwalls under wild breeding conditions generally select a diet providing about 28% protein (Krapu 1979). Shoveler, Blue-winged Teal, Lesser Scaup, Canvasback and Redhead hens consume a diet containing higher protein levels (Krapu 1979, Noyes & Jarvis 1985).

Inadequate nutrition adversely affects reproduction. Krapu (1979) pointed out the importance of a stable source of high quality foods during the nesting season. A 72% withdrawal of a nutritionally adequate control diet from Mallard hens during the laying cycle resulted in a 45% decline in the total number of eggs laid during the nesting season. Calcium deficient diets may also cause reduced egg production (Grau 1968). Calcium must come primarily from foods such as snails.

Diets with inadequate protein also cause hens to ovulate at irregular intervals, lengthening the period required to complete the clutch (Krapu 1974a). An extended laying period increases the risk of nest destruction. In addition, Krapu (1979) reported that diet quality influenced egg size and lipid reserves. These findings suggest that the quantity and quality of food reserves available during nesting can significantly affect the quantity of nutrient reserves available to ducklings at hatch. Nutrient reserves provide energy to the newly hatched ducklings and are crucial to their survival if adverse environmental conditions prevent them from obtaining adequate food in the initial few days after hatch. Animal matter in the diet of laying female dabblers was also necessary to assure adequate hatchability of eggs (Krapu & Swanson 1975, Krapu 1979).

In contrast , lipid needs in the diet are small because hens can synthesize most lipids or draw upon reserves. Lipids stored in the yolk are of great importance to the developing embryo and newly hatched duckling.

The nutritional demands of egg formation are met by the inclusion of high quantities of aquatic macroinvertebrates in the diet of laying hens. Table 2.27 summarizes the types and proportions of foods consumed by laying hens in their natural breeding habitat. A more detailed list of selected invertebrate prey is presented in Table 2.28. Aquatic insects, crustaceans and snails are highly selected, presumably because of their nutritional value, digestibility and availability. Although the composition of the animal portion of the diet of laying females varies among the duck species listed, all hens consumed more than 70% animal food. Diving species and the Northern Shoveler and Blue-winged Teal are almost totally dependent upon invertebrates.

Analyses of the nutrient composition of plant and animal foods consumed by female ducks breeding in the prairie pothole region confirm the high protein content of aquatic invertebrates (Table 2.29). All taxonomic groups analyzed contained 2-4 times the protein levels of plant foods. The essential amino acid composition of invertebrates fed upon by Pintails matched well with that of Pintail eggs (Krapu & Swanson 1975). Cladocerans and snails also are good sources of calcium. The invertebrate diet selected by laying hens is thus exceptionally high in protein and essential mineral nutrients.

The demand for invertebrate food changes with the reproductive status of the breeding hen. Animal foods comprised 56% of the Pintail diet during pre-laying, 77% during laying and 29% during post-laying (Krapu 1974b). For Gadwall hens the figures were 48%, 72% and 46% for the prelaying, laying and post-laying periods, respectively (Serie & Swanson 1976). Female Redheads consumed 25%, 77% and 49% animal food during the

Table 2.27. Percentage of major types of foods consumed by hens of certain species of wild ducks during the laying period in the prairie pothole region. Data are based upon volumetric measurements of esophaegeal contents[a].

	Percent Food Type						Percent Total		
Species	Gastropoda	Oligochaeta	Crustacea	Insecta	Seeds	Plant Parts	Animal	Plant	Reference
Dabblers									
Mallard	17	12	13	27[b]	25	3	72	28	Swanson *et al*. 1979 Swanson *et al*. 1985
Pintail	15	11	14	37	23	tr[c]	77	23	Krapu 1974b
Blue-winged Teal	40	1	14	44	1	--	99	1	Swanson *et al*. 1974
Gadwall (saline)	--	--	20	52	1	27	72	28	Serie & Swanson 1976
Gadwall (fresh)	4	tr	32	36	3	25	72	28	Swanson *et al*. 1979
Northern Shoveler	40	--	54	5	1	tr	99	1	Swanson *et al*. 1979
Divers									
Lesser Scaup	4	--	46	45	tr	1	98	2	[d]Bartonek & Hickey 1969a
Canvasback	66	--	--	26	tr	7	92	8	[d]Bartonek & Hickey 1969a
	26	--	--	52	2	16	78	22	[e]Noyes & Jarvis 1985

Table 2.27 (cont'd)

Species	Percent Food Type						Percent Total		Reference
	Gastropoda	Oligochaeta	Crustacea	Insecta	Seeds	Plant Parts	Animal	Plant	
Redhead	tr	--	--	80	4	15	81	19	[d]Bartonek & Hickey 1969a
	1	--	--	38[b]	2	21	77	13	[e]Noyes & Jarvis 1985
Ruddy Duck	22	1	5	67	5	--	95	5	Siegfried 1973

[a] Modified from Krapu 1979
[b] Remaining proportion of animal matter not categorized
[c] Trace
[d] Specific reproductive stage not reported
[e] Data from studies of breeding birds at the Ruby Lake National Wildlife Refuge, Nevada; based on percent dry weight

Table 2.28. Invertebrate prey selected by hens of wild ducks during their reproductive period.

Species	Selected Invertebrate Prey in Approximate Order of Importance	Reference
Dabblers		
Mallard	earth worms (Oligochaeta) aquatic snails (Lymnaeidae, Planorbidae) caddisfly larvae (Tricoptera) mosquito larvae (Culicidae) midge larvae (Chironomidae) predaceous diving beetle larvae (Dyticidae) dragon- and damselfly nymphs (Odonata) water boatman (Orixidae) clam shrimp (conchostraca) fairy shrimp (Anostraca)	Swanson *et al.* 1979, 1985
Pintail	midge larvae, pupae (Chironomidae) aquatic snails (Lymnaeidae) fairy shrimp (Anostraca) earthworms (Oligochaeta) water beetle larvae, adults (Coleoptera) clam shrimp (Conchostraea)	Krapu 1974b
Blue-winged Teal	aquatic snails (Lymnaeidae, Planorbidae, Physidae) midge larvae, adult (Chironomidae) scuds (Amphipoda) caddisfly larvae (Tricoptera) biting midge larvae (Ceratopogonidae) mosquito larvae (Culicidae) fairy shrimp (Anostraca) predaceous diving beetle larvae (Dyticidae)	Swanson *et al.* 1974 Swanson & Meyer 1977
Gadwall	midge larvae (Chironomidae) predaceous diving beetle larvae (Dyticidae) clam shrimp (Conchostraca) waterflea (Cladocera) caddisfly larvae (Tricoptera) seed shrimp (Ostracada) dragon- and damselfly nymphs (Odonata) water boatman (Corixidae)	Serie & Swanson 1976
Northern Shoveler	waterflea (Cladocera) aquatic snails (Lymnaeidae, Planorbidae) clam shrimp (Conchostraca) fairy shrimp (Anostraca) (Copepoda)	Swanson *et al.* 1979

Table 2.28 (cont'd)

Species	Selected Invertebrate Prey in Approximate Order of Importance	Reference
Divers		
Lesser Scaup	scuds (Amphipoda) midge larvae, pupae (Chironomidae) aquatic snails (Lymnaeidae) caddisfly larvae (Tricoptera) leeches (Hirudinea)	Bartonek & Hickey 1969a Dirschl 1969
Canvasback	aquatic snails (Lymnaeidae) mayfly nymphs (Ephimeroptera) caddisfly larvae (Tricoptera) midge larvae, pupae (Chironomidae) dragon- and damselfly nymphs (Odonata)	Bartonek & Hickey 1969a Noyes & Jarvis 1985
Redhead	caddisfly larvae (Tricoptera) fish eggs	Bartonek & Hickey 1969a Noyes & Jarvis 1985
Ruddy Duck	midge larvae, pupae (Chironomidae) aquatic snails (Gastropoda) scuds (Amphipoda)	Siegfried 1973

Table 2.29. Nutrient composition of certain plant and animal foods consumed by female dabbling ducks breeding in the prairie pothole region (from Krapu 1979).

Food Item		% Nutrient Composition (dry)					
Scientific Name	Common Name	Protein	Fat	NFE	Fiber	Ca	P
Plant							
Cladophoraceae [a]	green algae	16.0	0.2	41.3	22.4	2.9	0.6
Glyceria grandis [a]	mannagrass (caryopses)	6.0	1.4	76.1	7.9	0.3	0.5
Scolochloa festucacea [b]	whitetop (caryopses)	8.8	1.9	67.9	16.1	0.4	0.4
Triticum aestivum [b]	wheat (grains)	18.2	1.7	75.8	2.4	0.03	0.6
Echinochloa crusgalli [b]	barnyard grass (grains)	14.2	0.5	46.6	31.3	0.05	0.6
Beckmannia syzigachne [a]	sloughgrass (caryopses)	7.0	6.5	59.6	20.0	0.5	0.4
Animal							
Oligochaeta [b]	earthworms	60.2	7.7	12.3	0.4	0.2	1.0
Anostraca [a]	fairy shrimp	71.9	8.6	1.5	3.9	0.3	1.4
Cladocera [a]	waterfleas	31.8	1.5	10.9	7.3	11.8	1.2
Amphipoda [a]	scuds	47.0	5.9	16.5	8.4		
Diptera [b]	midges (larvae)	66.4	5.8	14.9		0.5	1.3
Gastropoda [b]	snails (without shell)	58.9	0.1	21.9	0.7	4.2	0.9
Gastropoda [a]	snails (with shell)	16.9	0.7	5.8	12.4	26.1	0.3

[a] Sugden (1973)

[b] Krapu and Swanson (1975)

same three reproductive stages and Canvasback hens changed from a pre-laying low of 8% invertebrates to 78% during both laying and post-laying (Noyes & Jarvis 1985).

There are seasonal trends in the diets of laying female ducks which reflect hydrological and phenological events that influence the availability of foods. For early laying female Mallards (April clutches), animal foods made up 68% of the diet (Swanson et al. 1985). Insect prey were primarily terrestrial forms drowned in early flooding. Anostraca, which mature early, were the most frequently consumed crustacean and terrestrial oligochaetes, which accumulate in shallow deposits due to spring rains, accounted for as much as 35% of the early nesters' diets. Hens laying in May consumed a proportion of animal food equivalent to early nesters but earthworms accounted for <1% of their diet. Snails and insects made up 25% and 26% of the diet, respectively. Diptera, Coleoptera and Trichoptera flies and larvae were heavily preyed upon as were Conchostraca and Anostraca. Late laying Mallards consumed a higher percentage (89%) of invertebrates than those hens laying earlier in the season. This may reflect increased nutritional demand as many late clutches are from renesting efforts. Odonata and Trichoptera each made up 20% of the late nesters diet and Cladocera replaced other crustaceans.

Early nesting Pintail hens, like Mallards, fed principally on Anostraca and oligochaetes in early spring (Krapu 1974a), whereas hens renesting during June fed primarily on midges and snails (Krapu 1974b). Mallards and Pintails begin their first nests in early spring when invertebrate foods are available only in ephemeral or temporary wetlands. As the nesting season progresses, seasonal wetlands become major sources of food. Late nesters and renesters forage on more permanent wetlands in late spring and early summer. Swanson et al. (1974) reported that the foods consumed by laying Blue-winged Teal hens varied among the aquatic habitat types used for foraging. Molluscs and insects dominated the diet on temporary and seasonal wetlands, whereas crustaceans, particularly large cladocerans and amphipods, were important prey in more permanent fresh wetlands and in some saline habitats. Gastropods also accounted for a large portion (38%) of Teal food on seasonal wetlands. A larger proportion of seeds are consumed on temporary and seasonal wetlands than in more permanent potholes.

The abundance of prey in seasonal wetlands may be influenced by local land-use practices. Throughout the Blue-winged Teal laying period, a higher density of invertebrate prey were found in undisturbed potholes than in those surrounded by cropland or fallow (Swanson et al. 1974). Food habits for breeding females foraging in natural and tilled wetland habitats are distinctly different. Pintails, the only species which readily use tilled wetlands, consumed 83% plant matter on tilled sites and 79% animal matter in non-tilled wetlands (Krapu 1974b). Caryopses of barnyard grass (Echinochloa crusgalli) made up more than 70% of the hen diet on the previously cultivated ponds.

Food resources are also influenced by hydrological conditions. During wet periods, Blue-winged Teals fed heavily on aquatic snails in seasonal wetlands, whereas during drought years, when most seasonal wetlands were dry, Teals fed principally on chironomid larvae in semi-permanent and permanent potholes (Swanson & Meyer 1977).

Some food preferences have been reported for dabbling and diving species. Bill morphology and feeding behaviour obviously influence prey selection. The distinctive bill morphology of the Northern Shoveler, incorporating finely spaced lamella, allows this species alone to sieve small zooplankton. This morphological feature explains the Shoveler's preference for cladocerans (Nudds & Bowlby 1984). The Ruddy Duck also has a specialized beak (Goodman & Fisher 1962) which it uses to strain large numbers of midge larvae from the soft mud bottom of ponds (Siegfried 1973). Prey selectivity may also be associated with feeding behaviour. Siegfried (1976) suggested that there were behavioural traits which segregated the food resources of Canvasbacks, Redheads, Scaups and Ruddy Ducks foraging sympatrically on potholes near Minnedosa, Manitoba.

Recent analysis by Nudds and Bowlby (1984) indicates that prey preferences may be based on selection of size rather than taxa. They suggested that the prey size selection by dabbling ducks is based on the lamellar density in the bill. Shovelers took the smallest prey followed by Green-winged Teals, Gadwalls, Blue-winged Teals and Widgeons with Pintails and Mallards consuming the largest prey. Only the Shoveler (70%), Gadwall (25%) and Blue-winged Teal (20%) took one-fifth or more of their total consumed energy in the form of prey <2.5 mm in length. Blue-winged Teals acquired another 60% of their energy from prey 2.6-12.5 mm in length. The Green-winged Teal and American Widgeon obtained about 75% of their energy intake from prey 2.5-7.5 mm in length. Mallards and Pintails took a large portion of their prey in sizes 2.6-12.5 mm in length, but these two species alone are capable of taking prey larger than 40 mm in length. Armstrong and Nudds (1985) demonstrated that the sizes of invertebrate prey were distributed such that the largest prey occurred in the shallowest microhabitats, whereas smaller prey were found in deep open microhabitats. They found that the foraging distribution of dabbling ducks was closely associated with the occurrence of prey in preferred size ranges.

Regardless of the reasons for preferential selection of invertebrate prey, a number of specific taxa are taken in proportions greater than their relative availability. Blue-winged Teals preferentially selected snails (Swanson et al. 1974) and Gadwall hens showed a high degree of selectivity for adult and larval dipterans as well as dysticid beetle larvae (Serie & Swanson 1976). Caddisfly larvae and dragonfly nymphs were the preferred foods of both Canvasback and Redhead females during laying (Noyes Jarvis 1985) while amphipods, primarily Gammarus and Hyalella, were preferentially selected by Lesser Scaup females during the breeding season

(Bartonek & Hickey 1969a). Larval and pupal chironomids were preferred by Ruddy Ducks (Siegfried 1973). In general, however, ducks are opportunistic in their food habits. This fact is exemplified by the correlations of high feeding activity with periods of maximum prey abundance and reports of unusually frenzied feeding activity associated with insect emergence peaks. For example, Sjöberg and Danell (1982) showed that feeding peaks for dabbling ducks coincided with chironomid emergence.

Although laying hens are opportunistic feeders and will switch prey with seasonal changes in invertebrate species abundance, laying efforts generally terminate when selected aquatic food resources start to decline (Pehrsson 1984). Reduced food availability also causes females to terminate renesting attempts (Swanson & Meyer 1977).

2.5.3.2 Ducklings

Although there are less data available on the diet of ducklings than on the diet of laying hens, evidence indicates a high dependence on animal food, primarily invertebrates, for periods of 1 to 7 weeks post-hatch. The digested protein content of food appeared to be the major factor influencing the growth rate of Mallard ducklings, particularly over the initial four days of growth (Street 1978). Diets of barley meal or seeds from aquatic plants were insufficient to maintain the ducklings: after 4-5 days of feeding solely on a plant food diet, when lipid reserves were depleted, birds lost weight and became weak. Street (op. cit.) concluded that a diet of greater than 50% animal protein was required to sustain a satisfactory growth rate in wild Mallard ducklings. He pointed out that, because growth rate is correlated with fat deposit (Marcström 1966) and plumage development (both important in temperature control), and with strength and activity, growth is roughly equivalent to survival. Therefore, fast growth of ducklings contributes to their success.

The contribution of major food groups to the natural diet of selected species of duckling foraging in the prairie pothole region is summarized in Table 2.30. Animal matter, predominantly invertebrates, composed an average of 91% of the diet of Mallard ducklings and 67% of the food of Pintail ducklings in the pre-fledgling stage. Although Gadwall and American Widgeon ducklings consumed an average of nearly 90% plant foods and only 10% invertebrates over the flightless growth period, most (80%) of the animal food was eaten in the critical first 20-day growth period (Sugden 1982). There was generally a greater dependency on animal prey in the diet of diving ducks than in dabbling species. Of the diving species listed, only the Redhead ducklings had an average intake of animal food substantially less than 90%. The 43% animal diet indicated for the Redhead may be biased as an inadequate number of birds <2 weeks of age were examined (Bartonek & Hickey 1969b). Where data was available on dietary consumption with age, class I ducklings (0-20 days) showed a much higher than average dependency on animal protein (Table 2.30). The period of dependency on

Table 2.30. Percentage of major foods consumed by ducklings prior to attaining flight in brood foraging areas in the prairie pothole region. Data are based on volumetric measurements of esophaegeal contents.

Species (Duckling Class)	Percent Food Type						Percent Total		Reference
	Gastropoda	Oligochaeta	Crustacea	Insecta	Seeds	Plant Parts	Animal	Plant	
Dabblers									
Mallard (I-III)	11	1	3	72[a]	9	--	91	9	Perret 1962
(I)							99	1	
Pintail (I-III)	36	--	4	26[a]	32	1	67	33	[b]Sugden 1973
(I)							>91	<9	"
Gadwall (I-III)	--	--	2	8	18	72	10	90	[b]Sugden 1973
(I)							>71	<29	"
American Widgeon (I-III)	2	--	--	9	13	76	11	89	[b]Sugden 1973
(I)							>89	<11	"
Divers									
Lesser Scaup (I-III)	16	1	53	26	2	2	96	4	[b]Sugden 1973
(I)							>98	<2	"
Canvasback (I-III)	15	--	--	79	1	5	94	6	[b]Bartonek & Hickey 1969b

Table 2.30 (cont'd)

	Percent Food Type						Percent Total		
Species	Gastropoda	Oligochaeta	Crustacea	Insecta	Seeds	Plant Parts	Animal	Plant	Reference
Redhead (I-III)	--	--	--	43	45	12	43	57	[c]Bartonek & Hickey 1969b
Ruddy (I-III) Duck	--	--	10	78	10	2	88	12	Siegfried 1973

[a] Remaining proportion of animal matter not categorized

[b] Data based on percent dry weight

[c] Only 5 of 49 ducklings were 2 weeks of age or younger, thus, proportion may not adequately represent the diet of young ducklings

invertebrates varies for duckling species (Table 2.31). Gadwall ducklings have a diet composed of more than 70% invertebrate foods for only the initial five days of growth and by three weeks of age take 5% or less invertebrate foods. In contrast, invertebrates dominate the diets of Widgeons, Pintails and Mallards for approximately the first 20 days of growth. Perret (1962) reported that juvenile Mallards in the Minnedosa, Manitoba, area were dependent on invertebrate food resources for a much longer period. Canvasbacks (Bartonek and Hickey 1969b) and Ruddy Ducks (Siegfried 1973) are highly dependent on invertebrate food resources for the entire pre-fledgling period.

Specific food preferences are summarized in Tables 2.32 and 2.33. Class I Mallard ducklings fed heavily on flying insects, particularly chironomids, as well as the larval and pupal stages of midges. Larvae of cased caddisflies (particularly Limnephilidae), aquatic snails (particularly Lymnaeidae), dragon- and damselfly naiads, water boatmen (Corixidae) and aquatic beetles (Hydrophilidae and Dytiscidae) were taken in moderate amounts (Perret 1962). Older Mallard ducklings selected more plant foods including grass, seeds, nutlets of Scirpus and seeds and tubers of Potamogeton (Chura 1961). Pintail ducklings chiefly ate surface invertebrates for their first five days and then consumed greater proportions of aquatic invertebrates and plant foods (Sugden 1973). Snails made up 36% by weight of the total diet but contributed only 10% of the gross energy intake (Sugden 1973). Midge larva and pupae composed 15% of the diet. Cladocerans were the only crustaceans eaten in significant amounts. Older Pintail ducklings consumed grain (9%), seeds of Puccinellia and nutlets of Eleocharis and Scirpus in substantial amounts (Sugden 1973).

Gadwalls ate primarily surface invertebrates immediately after hatching but switched to an exclusively plant diet by three weeks of age (Sugden 1973). Chironomid adults and larvae made up nearly one-third of the invertebrate food resources. Beetle larvae, particularly those taxa associated with macrophytes like Curculionidae and Haliplidae, accounted for most of the remainder of insect foods. Cladocerans made up 16% of the animal diet. Important plant foods included Potamogeton foliage (34%), green algae (19%), Beckmannia seeds and Lemna (Sugden 1973).

American Widgeon ducklings, like Gadwalls, fed widely on chironomid larvae and adults. Large numbers of ceratopogonid pupae were also consumed (Sugden 1973); in all, dipterans made up 48% of animal food consumed. Snails comprised 15% of animal diet. Potamogeton foliage and green algae were the main plant foods consumed: together they composed 65% of total diet (Sugden 1973).

Ducklings of diving species are predominantly invertebrate feeders throughout the flightless growth period. Amphipods were the most important food for Scaup ducklings (52% of their diet) (Sugden 1973). Chironomid larvae and pupae were also consumed in substantial quantities (15% of diet), particularly by ducklings in the first age class. Snails were preyed upon in appreciable amounts (16% of diet). Hemipterans, zygoptera naiads,

Table 2.31. Period of duckling growth during which invertebrates account for more than 70% of food consumed.

Species	Age Group (Days)	Reference
Mallard	0-18	Chura 1961
Pintail	0-20	Sugden 1973
Gadwall	0-5	"
American Widgeon	0-15	"
Lesser Scaup	0-50+	"
Canvasback	0-50+	Bartonek & Hickey 1969b
Redhead	[a]	"
Ruddy duck	0-50+	Siegfried 1973

[a] Insufficient age species dietary data, particularly for birds less than 2 weeks of age

Table 2.32. Invertebrate prey selected by flightless ducklings foraging in the prairie pothole region.

Species	Selected Invertebrate Prey in Approximate Order of Importance	Reference
Dabblers		
Mallard	true fly larvae, pupae, adults (primarily Chironomidae) caddisfly larvae (primarily Limnophilidae) aquatic snails (particularly Lymnaeidae) dragon- and damselfly nymphs (Odonata) water boatman adults, nymphs (Orixidae) beetle larvae, adult (Hydrophilidae, Dytiscidae)	Chura 1961 Perret 1962
Pintail	aquatic snails (Physidae, Lymnaeidae) midge larvae, pupae (Chironomidae) waterfleas (Cladocera)	Sugden 1973
Gadwall	midge larvae, adults (Chironomidae) beetle larvae (Curcullonidae, Dytiscidae, Halipilidae) water boatman adults, nymphs (Corixidae) waterfleas (Cladocera)	Sugden 1973
American Widgeon	midge larvae, adults (Chironomidae) biting midge pupae (Ceratopogonidae) aquatic snails (Lymnaeidae) water boatmen (Corixidae)	Sugden 1973
Divers		
Lesser Scaup	scuds (Amphipoda) aquatic snails (Lymnaeidae) midge larvae, pupae (Chironomidae) dragonfly naiads (Zygoptera) caddisfly larvae (Leptoceridae) beetle larvae (Haliplidae, Dytiscidae)	Bartonek & Hickey 1969a Sugden 1973
Canvasback	caddisfly larvae (Tricoptera) midge larvae (Chironomidae) aquatic snails (Lymnaeidae, Physidae) mayfly nymphs (Ephimeroptera)	Bartonek & Hickey 1969a,b

Table 2.32 (cont'd)

Species	Selected Invertebrate Prey in Approximate Order of Importance	Reference
Redhead	caddisfly larvae (Tricoptera) midge larvae (Chironomidae) dragon- and damselfly naiads (Odonata)	Bartonek & Hickey 1969a,b
Ruddy Duck	midge larvae, pupae (Chironomidae) scuds (Amphipoda) caddisfly larvae (primarily Limnephilidae)	Siegfried 1973

Table 2.33. Plant foods selected by flightless ducklings foraging in the prairie pothole region.

Species	Selected Plant Foods	Reference
Dabblers		
Mallard	nutlets *Scirpus*	Chura 1961
	seeds *Zancichellia*	
	seeds, tubers *Potamogeton*	
	seeds *Gramineae*	
Pintail	barley *Hordeum vulgare*	Sugden 1973
	seeds *Puccinellia*	
	nutlets *Eleocharis*	
	nutlets *Scirpus*	
Gadwall	algae *Cladophoraceae*	Sugden 1973
	foliage *Chara*	
	foliage *Potamogeton*	
	seeds *Beckmannia*	
	foliage *Lemna*	
American Widgeon	foliage *Potamogeton*	Sugden 1973
	algae *Cladophoraceae*	
	nutlets *Carex*	
	foliage *Lemna*	
Divers		
Lesser Scaup	oögonia *Chara*	Sugden 1973
	nutlets *Myriophyllum*	
Canvasback	oögonia *Chara*	Bartonek & Hickey 1969a,b
Redhead	caryopses *Scolochlon*	Bartonek & Hickey 1969a,b
	achenes *Scirpus*	
	oögenia *Chara*	
	nutlets *Potamogeton*	
Ruddy Duck	seeds *Scolochloa*	Siegfried 1973
	seeds *Lemna*	

leptocerid caddisfly larvae and haliphid and dytiseid beetles all represented greater than 1% of the food of Scaup ducklings (Sugden 1973). Ducklings of all ages ate only 4% plant food of which most was *Chara* oögonia and *Myriophyllum* nutlets (Sugden 1973).

The bulk of the diet of Canvasback ducklings consisted of caddisfly larvae (72%). Midge larvae were the second most widely selected prey making up 15% of duckling diet and snails, mayfly nymphs and odonate naiads composed the remainder of the animal portion of the diet (Bartonek & Hickey 1969a, b). The oögonia of *Chara* was the only plant food taken in significant quantities by juvenile Canvasbacks (Bartonek & Hickey 1969b).

The animal portion of the diet of Redhead ducklings was similar in composition to that of Canvasback ducklings. Caddisfly larvae made up nearly 40% of food intake (Bartonek & Hickey 1969b). Plant foods were of more importance to the Redhead than to ducklings of other diving species. Whitetop river grass seeds (*Scolochloa*) made up 44% of diet volume (Bartonek & Hickey 1969b) and achenes of *Scirpus*, oögonia of *Chara* and nutlets of *Potamogeton* were also consumed in appreciable quantities.

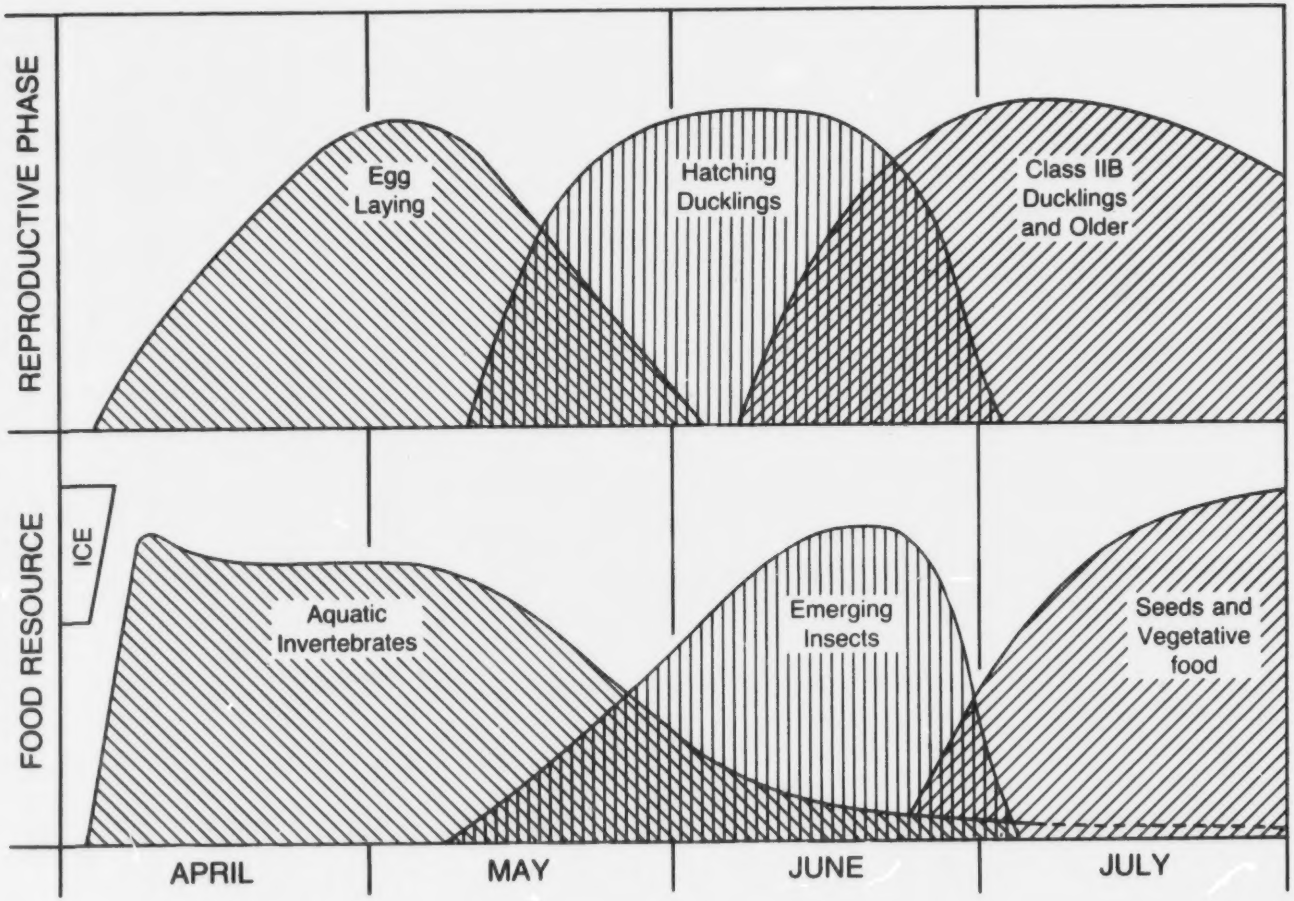
Food selection by ducklings, as with adults, depends on feeding behaviour and food availability. The inter-species similarity in duckling diet for the initial week post-hatch is paralleled by a similarity in feeding behaviour and feeding apparatus (Sugden 1973). The bill of newly hatched ducklings is relatively unspecialized. As a duckling grows its bill becomes more specialized. Sugden (1973, 1982) discussed the concomitant changes in feeding behaviour and diet with bill specialization in Pintail, Gadwall and American Widgeon ducklings. He also indicated that feeding changes with muscular and physiological development. The ability to digest plant foods may increase with age allowing more use of the available food resource.

Figure 2.10 shows the adaptation of reproductive phases to available food resources for Mallards in Sweden and stresses the importance of macroinvertebrate food availability to reproductive success (Pehrsson 1984). The general relationship between nutritional demands of laying hens and ducklings may be applied to early nesting dabblers in the pothole region of the Canadian prairie. The increased nutritional demand during the reproductive phase of the duck life cycle has critically linked breeding pairs and small ducklings to the protein-rich invertebrate food resources of their aquatic habitat.

2.5.3.3 Quantity of food resources

Dzubin (1969a) used the concept of habitat carrying capacity (defined as space) to assess the possible effects of population density on Mallard production. Pehrrson (1984) intimated that the carrying capacity concept should be applied to the influences of food resource availability

Figure 2.10. The relationship between food resources and critical periods of demand by laying hens and broods (from Pehrsson 1984).


REPRODUCTIVE PHASE
Egg Laying
Hatching Ducklings
Class IIB Ducklings and Older
FOOD RESOURCE
ICE
Aquatic Invertebrates
Emerging Insects
Seeds and Vegetative food
APRIL
MAY
JUNE
JULY

on duck reproductive success. Experimental quantification of natural food resource demands of breeding hens and broods is currently underway (G. Swanson, U.S.F.W.S. Northern Prairie Wildlife Research Center, pers. comm.), but is not well defined in the scientific literature. Bengston (1971) correlated reductions of 0.2 to 0.7 eggs in mean clutch size for eight duck species during a year when chironomid abundance was only 18% of normal standing crop. The standing crop of midge larvae in the Swedish lakes was 450 individuals m^{-2} in early June and 1,500 individuals m^{-2} in late July in the low year in comparison to normal values of that >2,500 m^{-2} in June and 25,000 m^{-2} in July. Krapu _et al_. (1983) found that midge densities averaging 673 larvae m^{-2} did not support significant nesting activity and considered levels of >10,000 larvae m^{-2} more favourable for waterfowl production. Brood movement in search of sufficient food has also been related to midge densities. Talent _et al_. (1982) reported that broods tended to emigrate from ponds where densities were <200 m^{-2} to ponds with larval abundances of >2,000 m^{-2}. Sedentary broods were generally found on ponds with 2,000-10,000 m^{-2} chironomid larvae.

2.5.4 Indices of reproductive success

To properly assess the impacts of environmental degradation on waterfowl in their breeding habitat, reproductive success must be measured. The study design should include assessment in a perturbed habitat and comparable assessment of success in an "ecologically equivalent" undisturbed environment. Studies of this type are, of necessity, intensive and long-term. Annual aerial survey data is adequate for trend analyses for more abundant species (e.g. Mallard) but generally lack the resolution and specificity required for a properly designed impact assessment. This section examines the measures of reproductive success which are key to testing the biological significance of pesticides on waterfowl resources.

Several indices of reproductive potential and reproductive or recruitment success have been reported in the waterfowl literature. Selected measures are defined and described in Table 2.34. The estimated number of breeding pairs or nests in an area is an index of reproductive potential. Nest success and number of broods produced are the initial measures of reproductive success reflecting environmental and ecological conditions. However, these indices do not account for brood survival through the flightless period and are therefore generally inadequate as a base for estimating population recruitment. Although recruitment success can be crudely estimated as the proportion of broods produced per breeding pair, the ultimate measure of recruitment (production) is the number of fledged juveniles per breeding pair.

Accurate estimation of the reproductive potential or welfare of duck populations is difficult because waterfowl censusing is inexact (Dzubin 1969b). Problems associated with the estimation of breeding pair, nest or brood density stem from a wide spectrum of factors. Weather, nesting

Table 2.34. Indices of reproductive success for ducks.

Reproductive Potential

1. Numbers of breeding pairs - number of pairs and lone males or small groups of males observed per unit area of habitat. Data is usually obtained from single or multiple ground beat-out census when the greatest proportion of species are in prenesting, laying or early incubation stages.

2. Nest density - number of nests per unit area of habitat. Data is obtained from systematic searches of upland, edge and emergent vegetation within a study area throughout the nesting period.

3. Clutch size - number of eggs per nest.

Reproductive Success

4. Nest success - proportion of nests which produce at least one duckling.

5. Hatch success - proportion of total eggs produced which hatch in a successful nest.

6. Number of broods produced - number of class II and III broods per unit area of habitat. Data is obtained from one or two censuses timed to measure early hatches and production of broods by late nesters and renesters.

7. Brood size at fledging - number of ducklings in class III broods.

Recruitment Success

8. Broods per pair - number of broods censused divided by the number of pairs censused.

9. Recruitment - number of fledged juveniles per breeding pair censused.

---

chronology and asynchronous nesting periods, and species and individual differences in daily activity patterns influence where and when ducks may be observed in various portions of their habitat. In addition, vegetative growth may restrict both bird and nest sitings and the high mobility of ducks (and broods of some species) makes the association of pairs (broods) with specific areas difficult. There are also unknown (or at least unquantified) observer-induced effects on reproducing ducks. Livezey (1980) reported that some nest abandonment resulted from nest disturbance and was most frequent during early egg laying. Also, avian predators may key in on searchers, thus increasing egg losses from censused nests (Strang 1980).

Bias problems also exist in some study results where the differences in the probabilities of finding successful and unsuccessful nests is not recognized. Most studies involve periodic searches. If nests are missed because they were initiated and destroyed between searches, the observed nest density would be biased downward and the observed nesting success biased upward (Miller & Johnson 1978). Mayfield (1961) recognized the bias which resulted from combining nest sighting data on clutches observed in various states of development as the probability of observing a successful hatch after the first sighting of the nest is high in mature clutches and low in nests where the first few eggs are being laid. He suggested a correction accounting for the exposure period (nest days).

Dzubin (1969b) presented an insightful analysis of problems in assessing breeding duck populations. He recommended 1) counting strategies specific to species; 2) corrections for biased sex ratios; 3) optimum census periods for early, intermediate and late nesters (minimum of three counts); and 4) counts done between 0800-1200 hours when pairs and drakes localize their activity. Dzubin also stressed the importance of supplementary data on the proportion of pairs which are non-breeding, the duration of the pair bond, the size of home range, as well as the distribution and ecology of potholes, to the proper interpretation of census data.

Brood survival and ultimately recruitment are the key variables in any assessment of breeding population success, but these measures are the most difficult to quantify. Recruitment censuses require much greater physical effort than breeding pair counts. With July vegetation in full bloom, broods are difficult to flush and their protective habits (hiding in thick vegetative cover) make counts and identification difficult (Smith 1971). The loss of whole broods of mobile dabblers to predators complicates production calculations.

2.5.4.1 Pair and nest density and nesting success

Although a number of published studies report various aspects of duck reproductive success in agricultural and non-agricultural areas, no direct assessment of recruitment differences has been undertaken. Data on breeding pair density, nest density and nest success from selected intensive studies are summarized in Table 2.35. All studies were conducted in the

Table 2.35. Nest density and nesting success for selected studies of communities of dabbling and diving ducks in specific habitats and under various environmental conditions in the prairie pothole region.

Region	Year(s)	Nesting Habitat	Breeding Pair Density pairs/ha	Nest Density nests/ha	Percent Nesting Success	Reference
North Dakota	1969-74	untilled upland	—	0.20	25	Higgins 1977
		annually tilled cropland	—	0.02	17	"
South Dakota	1971	idle cropland (predator reduction)	0.33	2.99	92	Duebbert & Kantrud 1974
		idle cropland (no predator reduction)	—	0.84	68	"
		active agricultural land (predator reduction)	0.17	0.12	85	"
		active agricultural land (no predator reduction)	0.20	0.14	51	"
North Dakota	1967-68	pasture grazed	--	0.42	14	Kirsch 1969
		pasture ungrazed	--	0.70	28	"
South Dakota	1971-73	undisturbed grassland/legume	—	0.76[a]	56[a]	Duebbert & Lokemoen 1976
Lousana, Alberta	1952-65	aspen parkland (pre-drought 1952-58)	0.66	0.12	44	Smith 1971
		aspen parkland (drought 1959-63)	0.49	0.07	35	"
		aspen parkland (post-drought 1964-65)	0.30	0.07	42	"
		aspen parkland (all years)	0.54	0.09	42	"
Redvers, Saskatchewan	1952-65	aspen parkland (all years)	0.26	0.12	36	Stoudt 1971
Oak Hammock, Manitoba	1974	wet sedge meadows	--	1.65	73	Oetting & Dixon 1975

[a] Dabbling ducks only

prairie pothole region and the nesting habitat was described in detail. Pair densities were lowest on actively cultivated agricultural land and higher on idle cropland and aspen parkland; nest density estimates were generally 15 to 50% of pair density estimates. The exception occurred when idle cropland with dense vegetative cover encouraged the nesting of dabblers not associated with local water bodies on the study site (Duebbert & Kantrud 1974). There are some obvious differences in nest density and nesting success associated with habitat type and condition. These indices were affected by tillage (Higgins 1977), grazing (Kirsch 1969),predator reduction (Duebbert & Kantrud 1974) and water conditions (Smith 1971, Stoud 1971). Nesting success varied from 14 to 92%. The long-term parkland studies indicate that average nesting success for that habitat was approximately 40%. Under current deteriorated conditions average nesting success is thought to be approximately half that measured by Smith and Stoudt (R. Greenwood, U.S.F.W.S. Northern Prairie Wildlife Research Center, pers. comm.).

Species-specific estimates of pair and nest density and nesting success are summarized in Table 2.36. The most complete data sets are for the aspen parklands from long-term studies at Lousana, Alberta (Smith 1971), and Redvers, Saskatchewan (Stoudt 1971). The Mallard has been the most intensively monitored. Average Mallard breeding pair densities for representative studies ranged from 0.1 to nearly 0.5 pairs per ha. Estimates from aerial census data for Alberta and Saskatchewan for the period 1955-73 indicated approximately 0.10 Mallards per ha in May (Pospahala et al. 1974). Nest densities were 2-5 times less than pair estimates and ranged from 0.03 to 0.15 Mallard nests per ha. Average nesting success rates were remarkably consistent in aspen parkland habitat (40%) and somewhat higher in the single grassland study (67%). Bellrose (1976) reported an overall average of 46% success for Mallards nesting in North America; Pintail nesting success was similar. More recent work has suggested that nest success rates for Mallards and Pintails are now far below these levels (G.L. Krapu, pers. comm.).

Only Blue-winged Teals approached Mallards in density (0.1 pairs per ha). Blue-winged Teal nests were found at a rate of 0.01 per ha and Bellrose (1976) calculated an overall average of 35% success rate for Blue-winged Teals and 31% for Green-winged Teals in North America. The remaining dabblers achieved substantially higher rates of nesting success. Gadwalls, American Widgeons and Northern Shovelers were less successful nesting in the aspen parklands than their counterparts in other regions (Bellrose 1976).

Pair and nest densities for Lesser Scaups and Canvasbacks were generally greater than 0.01 per ha. Scaups had a higher nesting success than Canvasbacks in Lousana but the opposite was true at Redvers. Nesting success for these species in the parklands was nearly equal to or greater than their respective species averages. The densities of Redheads and Ruddy Ducks were low in parkland study areas, however their estimated nesting success rates approximated species averages for North America.

Table 2.36. Pair and nest density and nesting success for duck species from selected studies in the prairie pothole region. Reported values are averages for the stated study period.

Species	Region	Years	Nesting Habitat	Pair Density nests/ha	Nest Density nests/ha	Percent Nesting Success	Reference
Dabblers							
Mallard	Lousana, Alberta	1952-65	aspen parkland	0.15	0.03	42	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.11	0.08	31, 44[a]	Stoudt 1971
	Kindersley, Saskatchewan	1956-58	mixed grassland	0.29	0.07	67	Dzubin & Gollop 1972
	Roseneath, Manitoba	1952-55	aspen parkland	0.49	0.15	36	"
						46[b]	Bellrose 1976
Pintail	Lousana, Alberta	1952-65	aspen parkland	0.03	0.01	43	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.03	0.01	40	Stoudt 1971
Blue-winged Teal	Lousana, Alberta	1952-65	aspen parkland	0.12	0.01	32	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.06	0.02	41	Stoudt 1971
						35[b]	Bellrose 1976
Green-winged Teal	Lousana, Alberta	1952-65	aspen parkland	0.03	0.01	9	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.01	0.01	26	Stoudt 1971
						31[b]	Bellrose 1976
Gadwall	Lousana, Alberta	1952-65	aspen parkland	0.02	0.01	31	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.01	0.01	52	Stoudt 1971
						68[b]	Bellrose 1976
American Widgeon	Lousana, Alberta	1952-65	aspen parkland	0.04	0.01	39	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.02	0.01	43	Stoudt 1971
						56[b]	Bellrose 1976
Northern Shoveler	Lousana, Alberta	1952-65	aspen parkland	0.01	0.01	50	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.01	0.01	47	Stoudt 1971
						59[b]	Bellrose 1976

Table 2.36 (cont'd)

Species	Region	Years	Nesting Habitat	Pair Density nests/ha	Nest Density nests/ha	Percent Nesting Success	Reference
Divers							
Lesser Scaup	Lousana, Alberta	1952-65	aspen parkland	0.09	0.02	47	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	<0.01	<0.01	41	Stoudt 1971
						43[b]	Bellrose 1976
Canvasback	Lousana, Alberta	1952-65	aspen parkland	0.01	0.01	36	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	0.01	0.01	65	Stoudt 1971
	Minnedosa, Manitoba	1962-72	aspen parkland	0.03	0.01	45, 59[a]	Stoudt 1982
						46[b]	Bellrose 1976
Redhead	Lousana, Alberta	1952-65	aspen parkland	0.01	<0.01	52	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	<0.01	<0.01	52	Stoudt 1971
						52[b]	Bellrose 1976
Ruddy Duck	Lousana, Alberta	1952-65	aspen parkland	0.01	<0.01	64	Smith 1971
	Redvers, Saskatchewan	1952-65	aspen parkland	<0.01	<0.01	6	Stoudt 1971
						70[b]	Bellrose 1976

[a] Adjusted for renesting effort

[b] Overall average for species in various geographical regions

Nesting success is highly dependent on habitat quality but is also influenced by other ecological factors. The Gadwall is the upland nesting species with the highest overall success rate (68%), largely due to its selection of dense nesting cover which greatly reduces the risk of predation. Egg predation pressure varies among duck species. The eggs of early nesters like Mallards and Canvasbacks are susceptible to avian predators (particularly crows) before spring vegetation has attained sufficient growth to restrict the visibility of nests. Crow populations of >30 birds per square hectare in prime duck habitat at Lousana inflicted substantial early nest damage (Smith 1971). Crow predation on nests of late nesters was markedly less, but losses to mammalian predators were high. Blue-winged Teals, which have a habit of nesting along fence rows, are vulnerable to the careful search habits of skunks and ground squirrels, which may explain their low nesting success rate. Scaup eggs are at high risk of mammal predation because of the Scaup's strategy of nesting near pothole edges where many mammals hunt. Martz (1967) reported that the highest nest losses of all duck species were near water. Late nesters generally have a higher success rate than early nesters due to the likelihood that they will select a stable nesting habitat with better cover development.

Intraspecific competition for habitat resources, forcing the use of marginal nest cover, and increased male harassment of nesting females are two probable reasons for reduced Mallard nest success (31.2%) in the densely populated areas of the prairie parklands (Bellrose 1979). Other differences in regional success of nesting waterfowl can be largely attributed to either habitat differences favoring populations in general or some species over others, or to a substantial reduction in habitat quantity or quality. For example, studies of island nesting habitats free of mammalian predators have found high nest success rates for most ducks (Duebbert et al. 1983). The probability of successful hatch of a clutch in regions of disturbed habitat such as intensively farmed lands is quite low. Overall nesting success rates for all species on an intensely farmed region of North Dakota was 21% (Higgins 1977), a figure well below the 36% and 42% success rate for all nesting species at Redvers and Lousana parklands, respectively (Stoudt 1971, Smith 1971). It is worth noting again that present conditions in much of the pothole region may be contributing to much lower nesting success rates than the averages cited above.

2.5.4.2 Clutch size and percent hatching success

Although average clutch size is species-specific, the range of the mean number of eggs per nest varies only by three eggs among the divers and dabblers listed in Table 2.37. Gadwalls, Blue-winged Teals and Redheads lay the greatest number of eggs and Pintails and Ruddy Ducks lay the fewest.

Nesting effort is a function of water condition and age of the hen (Cowardin et al. 1985). For all species, clutches laid early during the reproductive period are larger than those laid later. Dzubin and Gallop

Table 2.37. A summary of clutch size and percent hatching success data for duck species in the prairie pothole region. Reported values are averages for the stated study period.

Species	Average Clutch Size		Region	Years	Percent Hatching Success	Reference
	Overall[a]	Specific (Late Nesting/Renesting)				
Dabblers						
Mallard	9.0	7.8	Lousana, Alberta	1953-65	--	Smith 1971
		8.4	Redvers, Saskatchewan	1952-65	93	Stoudt 1971
		9.1 (7.7)	Kindersley, Saskatchewan	1956-58	93	Dzubin & Gollop 1972
		8.6 (7.2)	Roseneath, Manitoba	1952-55	93	"
					93[b]	Bellrose 1976
Pintail	7.8	7.7	Lousana, Alberta	1953-65	--	Smith 1971
		7.9	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
					88[b]	Bellrose 1976
Blue-winged Teal	9.8	10.5	Lousana, Alberta	1953-65	--	Smith 1971
		8.9	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
Green-winged Teal	8.6	9.4	Lousana, Alberta	1953-65	--	Smith 1971
		9.6	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
Gadwall	10.0	8.5	Lousana, Alberta	1953-65	--	Smith 1971
		10.0	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
					90[b]	Bellrose 1976
American Widgeon	8.5	8.9	Brooks, Alberta	1953-57	--	Keith 1961
		7.6	Lousana, Alberta	1953-65	--	Smith 1971
		9.0	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
Northern Shoveler	9.4	10.7 (9.4)	Brooks, Alberta	1953-57	--	Keith 1961
		9.3	Lousana, Alberta	1953-65	--	Smith 1971
		10.1	Redvers, Saskatchewan	1953-65	--	Stoudt 1971
					93[b]	Bellrose 1976

Table 2.37 (cont'd)

Species	Average Clutch Size		Region	Years	Percent Hatching Success	Reference
	Overall[a]	Specific (Late Nesting/Renesting				
Divers						
Lesser Scaup	9.0	8.6	Lousana, Alberta	1953-65	--	Smith 1971
		9.9	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
					93[b]	Bellrose 1976
Canvasback	7.9	7.7	Lousana, Alberta	1953-65	--	Smith 1971
	9.5 (non-parasitized nests)	8.2	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
		8.2	Minnedosa, Manitoba	1963-72	80	Stoudt 1982
Redhead	11.1 (parasitic laying hens)	10.0	Lousana, Alberta	1953-65	--	Smith 1971
		9.3	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
					51[b]	Bellrose 1976
Ruddy Duck	8.0	6.6	Lousana, Alberta	1953-65	--	Smith 1971
		8.8	Redvers, Saskatchewan	1952-65	--	Stoudt 1971
					70[b]	Bellrose 1976

[a] Data on average clutch size for individual species is from Bellrose 1976

[b] Overall average for species in various geographical regions

(1972) reported that Mallards in the pothole region of Roseneath, Manitoba, had an average clutch of 8.6 eggs prior to May 25 and 7.2 eggs after that date. Smaller clutches late in a species nesting period may be due to renesting stress (Coulter & Miller 1968) or to hen immaturity associated with a late nesting strategy. Stoudt (1971) reported that the smallest clutch size for Mallards in his 13 years of study was due to adverse weather causing loss of most early nests and a massive renesting effort. There is also some evidence that crowding may decrease the number of eggs laid, particularly during those periods when the total number of breeding ducks is greatly expanded (Smith 1971). The clutch size of Canvasbacks can be reduced if nests are parasitized by Redhead laying. Erickson (1948; cited in Bellrose 1976) reported 9.9 eggs in unparasitized Canvasback nests, but 7.7 Canvasback eggs in nests with some Redhead eggs. Intrusion by laying Redheads is a major cause of Canvasback nest abandonment (Bellrose 1976).

Clutch size can vary regionally or annually within a region depending upon the stress on breeding birds. Fluctuations in predator populations, changes in predator species and habitat alterations such as removal of vegetative cover have been cited as causes of substantial changes in clutch size (Smith 1971). In general, clutch size is greater in predator-free habitat. For example, more than 80% of Mallard and Gadwall clutches on Miller Lake Island, which is free of mammalian predators, were larger than the average clutch size for these species (Duebbert _et_ _al_. 1983). Habitat destruction may leave hens and nests more open to predators, thus decreasing the laying effort and increasing the likelihood of some egg destruction. Hens of most species do not accept the loss of more than a quarter of a clutch and may abandon the nest if discovered by a predator. Field burning or plowing in preparation for planting may directly destroy early nests increasing renesting stress and reducing the number of eggs per nest. Pintail nests are particularly vulnerable to agricultural activity.

The average number of undisturbed eggs which fail to hatch in a successful nest is very low for most species (see Table 2.37). An obvious exception to this generalization is the Redhead, which had only a 51% hatch rate for eggs in successful nests. The unusually high proportion of unhatched eggs probably indicates the extent to which intruder hens laid eggs after the host hen began to incubate (Bellrose 1976).

2.5.4.3 Brood success and recruitment

The successful fledging of hatched broods depends primarily on predation losses (Section 2.5.2) and food availability (Section 2.5.3). Recent declines in Mallard populations have been explained by low recruitment in pothole areas and high mortality during the summer (Cowardin _et_ _al_. 1985).

Indices of brood production and recruitment success estimated for selected long-term studies in the Canadian pothole region are presented in Table 2.38. The number of broods produced per pair for Mallards ranged from 0.26 to 0.70. Average Mallard brood size at fledging ranged from 4.9 to 6.1

Table 2.38. Measures of brood and recruitment success for duck species from selected long-term studies in the prairie pothole region. Reported values are averages for the stated study period.

Species	Region	Years	Broods per Pair	Brood Size at Fledging	Young per Pair	Reference
Dabblers						
Mallard	Lousana, Alberta	1953-65	0.31	5.1	1.6	Smith 1971
	Redvers, Saskatchewan	1952-65	0.33	6.0	2.8	Stoudt 1971
	Kindersley, Saskatchewan	1956-58	0.70	4.9	1.0	Dzubin & Gollop 1972
	Roseneath, Manitoba	1952-55	0.34	6.1	2.4	"
	Caron, Saskatchewan	1950-75	0.26	-	-	Leitch & Kaminski 1985
Pintail	Redvers, Saskatchewan	1952-65	0.23	5.1	2.0	Stoudt 1971
	Caron, Saskatchewan	1950-75	0.35	-	-	Leitch & Kaminski 1985
Blue-winged Teal	Lousana, Alberta	1953-65	0.41	7.8	3.2	Smith 1971
	Redvers, Saskatchewan	1952-65	0.31	7.7	3.2	Stoudt 1971
	Caron, Saskatchewan	1950-75	0.44	-	-	Leitch & Kaminski 1985
Gadwall	Caron, Saskatchewan	1950-75	0.22	-	-	Leitch & Kaminski 1985
American Widgeon	Lousana, Alberta	1953-65	0.47	7.0	2.7	Smith 1971
	Caron, Saskatchewan	1950-75	0.30	-	-	Leitch & Kaminski 1985
Northern Shoveler	Caron, Saskatchewan	1950-75	0.37	-	-	Leitch & Kaminski 1985
Divers						
Lesser Scaup	Lousana, Alberta	1953-65	0.16	-	-	Smith 1971
	Caron, Saskatchewan	1950-75	0.22	-	-	Leitch & Kaminski 1985
Canvasback	Lousana, Alberta	1953-65	0.62	4.5	2.8	Smith 1971
	Redvers, Saskatchewan	1952-65	0.65	5.4	3.4	Stoudt 1971
	Minnedosa, Manitoba	1963-72	0.67	5.3	2.6	Stoudt 1982

birds and estimated recruitment varied between 1.0 to 2.4 young per breeding pair. Although production was highest at the Kindersley grassland study area, the estimated survival of newly hatched broods from nest to water was lowest at this site, only 48% (Dzubin & Gollop 1972). The high loss of entire broods was associated with the great distances that hens nested from ponds in this grassland habitat. The majority of broods had to travel 200 m or more to water after hatching (Dzubin & Gollop 1972). In a recent study in North Dakota, Cowardin et al. (1985) estimated Mallard recruitment to be 0.27 young females fledged per female in the breeding population. This low rate may be indicative of the problems Mallard are having in some parts of their breeding range.

Pintails produced, on average, fewer broods per pair than did Mallards at Redvers and recruited fewer juveniles (2.0 per pair). Because of their use of open habitat and their extreme mobility, Pintails sustained high brood mortality (Stoudt 1971). Blue-winged Teals produced 0.31 to 0.44 broods per pair and were more successful than either Mallards or Pintails in bringing broods to successful fledge: recruitment of juvenile Teals at both Lousana and Redvers averaged 3.2 young per pair. The high brood survival rate of Blue-winged Teals tends to offset their low nesting success.

Brood success for Gadwalls and American Widgeons was intermediate between the low rates recorded for Pintail and the high rates of Blue-winged Teals. Bellrose (1976) estimated that 2.2 Gadwall young per pair were produced for North American populations. A much higher production estimate (4.0) of young per pair was reported for Waterhen Marsh, Saskatchewan (Hines & Mitchell 1983). At Lousana, an estimated 2.7 young per pair were recruited into the fall Widgeon population (Smith 1971). The production of Shoveler broods per pair was 0.37 at Caron, Saskatchewan (Leitch & Kaminski 1985).

Although Canvasback broods are highly mobile, the loss of whole broods is probably a rare event (Bellrose 1976). The low success of Canvasback broods at Minnedosa, Manitoba, may be attributed to raccoon predation. Estimated Canvasback production in Alberta and Saskatchewan parkland potholes ranged from 2.8 to 3.4 young per pair (Smith 1971, Stoudt 1971).

Lesser Scaups had the lowest brood production estimated in these selected studies: between 0.16 and 0.22 broods were produced per pair. No estimates of brood survival were provided.

2.5.4.4 Models and recruitment estimates

Although recruitment data is key to assessing population success on duck breeding grounds, the estimation of waterfowl recruitment has not received the same attention as has the estimation of survival rates. Recruitment can be estimated either by 1) direct measurement of breeding populations, number of broods produced and brood size at fledging; or 2) nesting success information in conjunction with knowledge of renesting effort and brood survival (Dzubin & Gollop 1972). Cowardin and Johnson (1979) discussed the mathematics and assumptions in modelling recruitment. They defined recruitment (R) as the number of females fledged per hen. Recruitment can be calculated by the direct method:

$$R = IB / 2$$

where I is the ratio of the number of fledged broods to the number of breeding hens; B is the average size of fledged broods; and 2 accounts for an even sex ratio in broods. Cowardin and Johnson (1979) suggested that brood sizes based on class III estimates are frequently too large because these are measured from hatch counts of early nesters producing larger broods than late nesters (Blatt 1976). They recommended using class I count data and applying survival rate calculations.

Recruitment can alternatively be calculated from the hen success (H) model:

$$R = HZB / 2$$

where H is the probability that an individual hen will hatch a nest in one of her attempts; and Z is the survival of broods from hatch to time of brood census (class II). Cowardin and Johnson (1979) estimated average Z values of 0.7 for Mallard broods in the Canadian parklands. Z values may vary greatly among years and areas. Hen success not available from direct census can be estimated from the probabilty of a nest hatching (P) according to Miller and Johnson (1978):

$$H = Pe^{(1-P)^2}$$

assuming that nesting effort generally decreases as the breeding season progresses and that the probability that a hen will attempt to renest is inversely related to the probability its current clutch will hatch (i.e. the later during laying or incubation that a nest is destroyed, the less likely a hen will renest).

The indirect calculation model provides an estimate of recruitment in excellent agreement with the direct method with data from both the Smith (1971) and Stoudt (1971) studies.

As many studies of duck reproduction have failed to collect adequate data to estimate recruitment, only woefully incomplete baseline and comparative data sets are available. Current studies by personnel from the U.S.F.W.S. and CWS on factors influencing the productivity of upland nesting ducks in the prairie region may help further define the quantitive relationship between habitat factors and duck production (R. Greenwood, U.S.F.W.S. National Prairie Wildlife Research Center and K. Brace, CWS Migratory Birds, pers. comm).

CHAPTER 3

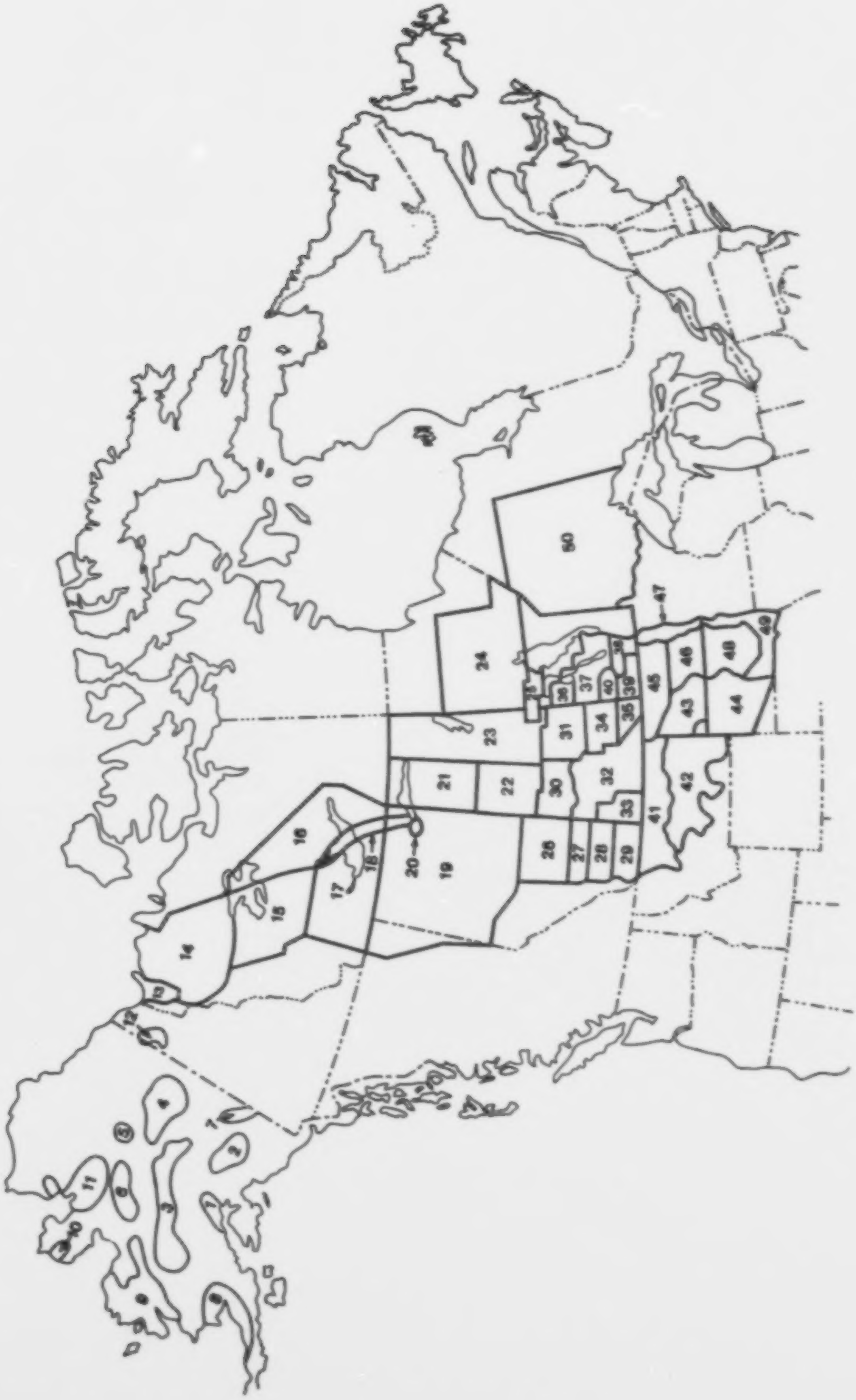
WATERFOWL HABITAT DISTRIBUTION AND AGRICULTURE OVERLAP

The previous chapter addressed the qualitative aspects of waterfowl and prairie slough ecology. In order to assess the risk associated with pesticide impact on the waterfowl resource of western Canada, it is necessary to provide a quantitative account of the waterfowl resource as well as an estimate of the spatial overlap between waterfowl production and agricultural land use practices. Although the impact of agriculture on waterfowl has long been the focus of attention of waterfowl biologists, there has not, to date, been a published analysis of the overlap between agricultural land use and waterfowl nesting with enough detail to permit the assessment of the potential impact of any one agricultural chemical control practice, much less the combined effect of pesticides, on the waterfowl resource. The main objective of this chapter is to present the waterfowl and agricultural data necessary for such an analysis. We have relied to a large extent on the unpublished work of Patterson (1978) who first tried to place the entire problem of overlap between waterfowl and agriculture into perspective by matching waterfowl population surveys with agricultural and waterfowl land capabilities. Using a slightly different approach, Boyd (1985) explored the broad scale impact of agriculture on waterfowl by correlating waterfowl numbers with changing land practices on a provincial scale. Our approach is not to seek a functional relationship between waterfowl and agriculture at that level but rather to break down the waterfowl data on a finer scale and by crop type so as to eventually be able to isolate the possible impact of pesticides from other agricultural disturbances. A secondary aim is to demonstrate the importance of the prairie pothole region of Canada to North American waterfowl production and illustrate some key points concerning the distribution of various species within this habitat. In addition, the basics of waterfowl surveys and the problems inherent in the quality and scope of the data are discussed. Where relevant, raw information has been provided (usually in appendix form) so as to allow for any future hazard determinations by resource managers and impact evaluators.

3.1 Waterfowl Numbers and Distribution

Since 1955, the United States Fish and Wildlife Service (USFWS) and the CWS have systematically monitored duck populations throughout the Canadian and U.S. prairies as well as in a limited section of their northern breeding habitats such as the open and closed boreal forest, taiga and arctic steppe. The areas surveyed are divided into a number of strata based partly on geography and partly on the abundance of the most common waterfowl species - primarily the Mallard (Figure 3.1). The assignment of strata to specific vegetation zones and the sizes of the strata are given in Table 3.1. Assigning strata to vegetation zones is rather arbitrary; we have followed the interpretation of Bellrose (1979) which differs slightly from Boyd (1985). The latter excludes strata 30, 31 and 38 from the parkland zone.

Figure 3.1. Waterfowl strata utilized for annual survey purposes.



1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31
32
33
34
35
36
37
38
39
40
41
42
43
44
45
46
47
48
49
50

Table 3.1. The vegetation form and surface area of waterfowl strata.

Vegetation Form	Stratum	Surface Area (km^2)
Parklands (Canadian)	26	68,500
	30	48,096
	31	54,612
	34	34,094
	38	14,643
	40	11,748
Total		231,693
Mixed Prairie (Canadian)	27	29,199
	28	33,385
	32	98,192
	35	23,424
	39	16,970
Subtotal		201,170
Mixed Prairie (U.S.)	45	68,958
	46	36,876
	48	63,685
Subtotal		169,519
Mixed Prairie (Total)		370,689
Shortgrass Prairie (Canadian)	29	34,278
	33	29,383
Subtotal		63,611
Shortgrass Prairie (U.S.)	41	85,216
	42	105,562
	43	51,372
	44	70,701
Subtotal		312,851
Shortgrass Prairie (Total)		376,512
Tallgrass Prairie	W. Minnesota	94,534
	N. Dakota 47	20,256
	S. Dakota 49	40,999
	N. Nebraska	56,980
Total		212,769
PRAIRIE TOTAL		1,191,663

Aerial surveys are conducted annually in May, to determine the breeding populations of each species, and in June/July, primarily to obtain an estimate of breeding success (brood counts) although the broods are not speciated. The surveys consist of transects, 0.4 km by approximately 29 km strips, typically flown east-west at an altitude of about 46 m. The pilot and observer each count ducks and ponds (and broods in June) on their respective sides of the aircraft (Bellrose 1976). Ground surveys are also conducted on approximately 10% of these air transects to determine visibility correction factors for each species in each ground cover type. The correction factors are then applied to all transects of that type. The total breeding population size for any stratum is extrapolated on an area basis from the adjusted transect counts. A recent detailed description of the survey can be found in Benning (1976) with a discussion of its limitations in Bowden (1974) and Martin _et al_. (1979). For the purposes of this section, the population estimates adjusted for visibility have been accepted (Appendices 3.1 to 3.4). In Appendix 3.1 the estimated breeding populations of each species surveyed for each stratum are presented. These estimates were averaged over a fourteen year period between 1970 and 1983. The estimated total prairie duck population (excluding coots) is about 22.3 million birds. These average population estimates are further broken down into Canadian and U.S. Prairie subtotals (Appendix 3.2), major geo-political area subtotals (Appendix 3.3) and vegetation zone (as defined in Chapter 2) subtotals (Appendix 3.4). It should be noted that the assignment of strata to a single vegetation zone (Table 3.3 and Appendix 3.4) leads to serious errors when estimating the number of breeding ducks by vegetation zone (J.B. Millar, pers. comm.). Fortunately the concept of vegetation zone is not crucial to our hazard evaluation but this caveat should be kept in mind in the discussion which follows. In all of these appendices, the total surveyed population (or common denominator) refers to totals obtained for all 50 strata as defined in Figure 3.1.

There are no direct surveys by which to assess the relative contribution of the annually surveyed strata to the whole of the North American duck population. However, a recent estimate of the total North American breeding duck spring population was given as 61.8 million (Anonymous 1985). This means that the total surveyed population (within the 50 strata) accounts for roughly 70% of the North American breeding duck population and that the Canadian prairies account for approximately 26% of the total North American breeding population. The importance of the prairies varies between species. Dabbling ducks are more numerous than diving ducks on the Canadian prairies, comprising about 84.4% of the breeding duck population (Table 3.2). The three most abundant species are the Mallard, Pintail and Blue-winged Teal. Population levels for these species over the past fourteen years have averaged approximately 4.0, 2.8 and 2.7 million, respectively. Of the eight remaining abundant duck species, the Northern Shoveler, American Widgeon, Scaup (primarily lesser) and Gadwall have population means ranging from 1 to approximately 1.1 million and the Green-winged Teal, Redhead, Canvasback and Ruddy Duck

Table 3.2. Population estimates of ducks and American Coot in the Canadian prairies compared to the total surveyed population levels (strata 1-50)[a] and to current Canadian and U.S. population levels of the same species[b].

Rank	Species	Population in Canadian Prairies (from Appendix 1)	Percent of Total Surveyed Population Present in Canadian Prairies	Percent of Total Surveyed Population Present in Combined Canadian and U.S. Prairies
1	Mallard (*Anas platyrhynchos*)	3,983.2	51.1	67.4
2	Pintail (*Anas acuta*)	2,826.6	49.4	65.7
3	Blue-winged Teal (*Anas discors*)	2,746.6	55.6	87.9
4	American Coot (*Fulica americana*)	1,445.3	59.0	86.4
5	Northern Shoveler (*Spatula clypeata*)	1,145.8	53.6	80.7
6	American Widgeon (*Mareca americana*)	1,092.7	33.1	47.2
7	Scaup (primarily Lesser) (*Aythya affinis*)	1,093.4	14.4	16.9
8	Gadwall (*Anas strepera*)	968.0	60.7	93.8
9	Green-winged Teal (*Anas carolinensis*)	678.2	28.2	36.1
10	Redhead (*Aythya americana*)	452.9	57.1	87.7
11	Canvasback (*Aythya valisneria*)	346.6	59.6	67.5
12	Ruddy Duck (*Oxyura jamaicensis*)	312.4	48.7	88.5
13	Bufflehead (*Bucephala albeola*)	103.2	11.7	12.1
14	Ring-necked Duck (*Aythya collaris*)	74.2	12.2	16.8
15	Goldeneye (*Bucephala clangula*)	64.6	8.5	8.7
16	Scoters (*Melanitta* sp.)	24.5	1.7	1.7
17	Mergansers (*Mergus* sp.)	21.0	3.3	3.6
18	Black Duck (*Anas rubripes*)	0.7	0.8	0.8
19	Oldsquaw (*Clangula hyemalis*)	.1	-	-
20	Eiders (*Somateria*, *Polysticta*)	.1	-	-
Total dabbling ducks[c]		13,441.8	48.0	67.9
Total diving ducks		1,967.1	16.2	21.2
Total ducks		15,934.6	36.7	51.3

[a] From Appendix 1 - estimates based on 14-year averages for each species over the period 1970-1983

[b] From Couch 1981

[c] 84.4% of Canadian Prairie population

Table 3.3. Comparison of breeding duck densities in the vegetation zones of each prairie province: 1960-69[a] versus 1970-83[b].

PROVINCE dominant vegetation zone	Vermeer (1972) strata	Current strata	Years	Number of Breeding Individuals/km^2										
				Mallard	Pintail	Gadwall	American Widgeon	Northern Shoveler	B-w Teal	G-w Teal	Redhead	Canvas-back	Scaup	Total
ALBERTA Parkland	27	26	60-69	12.2	4.5	3.2	3.7	2.9	6.6	2.2	1.2	1.0	3.6	41.1
			70-83	11.6	5.6	2.8	2.6	2.6	7.1	2.8	1.7	0.9	5.1	42.9
Mixed grassland	26	27,28	60-69	5.9	7.2	2.0	2.4	2.7	2.9	0.6	1.0	0.2	1.3	26.3
			70-83	5.2	8.3	1.6	1.9	2.5	3.1	0.9	0.4	0.2	1.5	25.6
Shortgrass	28	29	60-69	4.9	6.5	0.9	1.5	1.5	1.2	0.2	0.6	0.1	1.2	18.6
			70-83	3.9	7.0	1.0	1.6	1.8	1.7	0.5	0.3	0.1	1.0	18.9
SASKATCHEWAN Parkland	20	34	60-69	7.5	3.1	1.0	1.3	1.2	8.6	0.7	0.2	0.4	0.6	24.6
			70-83	9.6	4.5	1.5	1.7	2.0	7.6	2.2	0.8	1.0	1.3	32.3
Parkland	21	30	60-69	8.0	3.8	1.3	3.3	1.9	3.3	1.5	2.2	1.3	1.7	28.4
			70-83	10.1	4.2	1.9	3.1	2.2	5.2	0.9	1.1	1.3	2.9	32.9
Mixed grassland	19	32	60-69	5.3	4.9	2.2	2.3	2.1	2.8	1.3	0.2	0.3	0.7	22.1
			70-83	7.9	7.1	2.9	3.2	2.3	5.1	0.7	0.6	0.5	1.6	31.8
Shortgrass	23	33	60-68	4.2	4.6	1.5	2.0	1.5	1.8	1.0	0.1	0.2	0.4	17.4
			70-83	4.3	4.9	1.0	1.6	1.2	1.8	0.3	0.2	0.1	0.8	16.2
MANITOBA Parkland & mixed grassland	24	39,40	60-69	5.8	2.2	1.7	1.5	2.0	9.7	1.5	1.1	1.2	2.4	29.1
			70-83	6.2	2.8	1.5	0.8	1.7	9.7	1.9	1.4	1.1	2.6	29.8

[a] From Vermeer 1972
[b] From Appendix 3.1

range from 0.3 to 0.7 million. Eight other species are enumerated in the survey but their populations in the Canadian prairies are rarely higher than 0.1 million; 13% or less of their total surveyed breeding population utilize this habitat (Table 3.2). It should be noted that the standard deviations for these fourteen-year population means within a specific stratum can range from a low of 20% for a widely distributed and abundant species such as the Mallard, to in excess of 100% for some of the less numerous species (Appendix 3.1). Therefore, population differences of a few tenths of a million may not indicate a significant difference in the rank of one species over another.

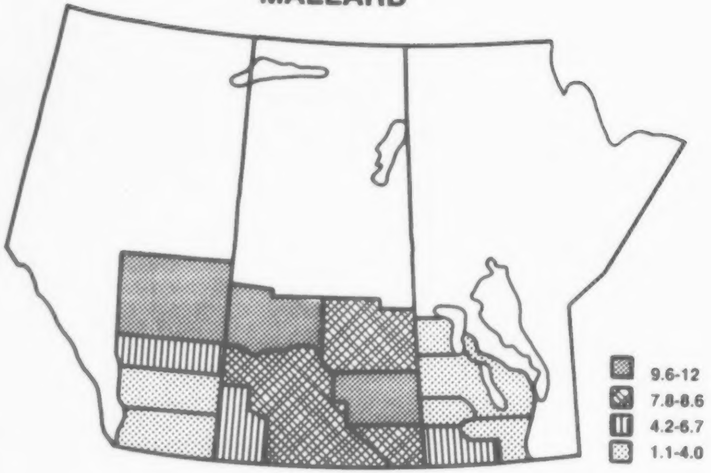
Of the eleven most abundant species, all but three have over 65% of their surveyed breeding population in the parklands and associated Canadian and U.S. grasslands indicating the very great importance of these habitats to continental duck populations. The three species which are the exception (American Widgeon, Scaup and Green-winged Teal) each have a large proportion of their breeding population nesting in the southern boreal forest or on northern river deltas such as the Mackenzie (Bellrose 1976). The American Coot, which is also surveyed annually on the waterfowl strata, is heavily dependent on the prairies with 86% of its breeding stock nesting there. Its fourteen-year population mean is over 1.4 million, which places it behind the three most abundant dabbling ducks in rank.

When the combined Canadian and U.S. prairie is considered, the mixed grassland area, on account of its sheer size in the U.S., contains the larger proportion of the total breeding duck population relative to the parklands or short and tallgrass zones. This balance shifts in favour of the parklands if only the Canadian habitat is considered. The parklands support higher densities of many species (Figure 3.2). Of note, the parklands of Alberta (Str. 26) have the highest densities of breeding Mallard, Gadwall, American Widgeon, Shoveler, Green-winged Teal and Lesser Scaup of the entire prairie region. However, although the grasslands of southern Saskatchewan are a less reliable habitat than the parklands for waterfowl breeding, they can, in wet years, produce phenomenal numbers of broods (Lynch 1984) and are, therefore, very important to overall waterfowl production. The mixed grass prairie of southern Alberta is especially important to the Pintail.

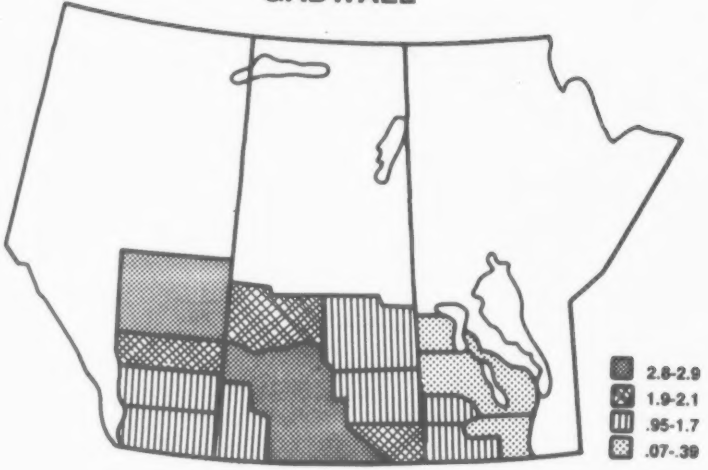
The distribution of the major duck species in the Canadian prairies has similarly been summarized by Vermeer (1972) for the period 1960-69. Although the exact aportioning of the strata was slightly different then, the area surveyed was the same and a direct comparison can be made between the two time periods (1960-69 versus 1970-83) for most of the prairie region (Table 3.3). In most cases these multi-year averages are in close agreement but they obscure important year to year fluctuations as well as a general decline in population numbers which has become apparent in recent years. Specifically, numbers of Mallards, Blue-winged Teal and Pintail are presently at an all-time low in the prairies (Brace & Caswell 1985). The Mallard data is especially worrysome since they are considered to be the most dependable of the survey (F.G. Cooch, pers. comm.). Several explanations for these declining stocks have been advanced: increased hunting pressure (primarily in the U.S.), periodic drought in the southern

Figure 3.2. Mean breeding duck densities (ducks/km^2) averaged over the 1970-1983 period for strata 26 to 40.

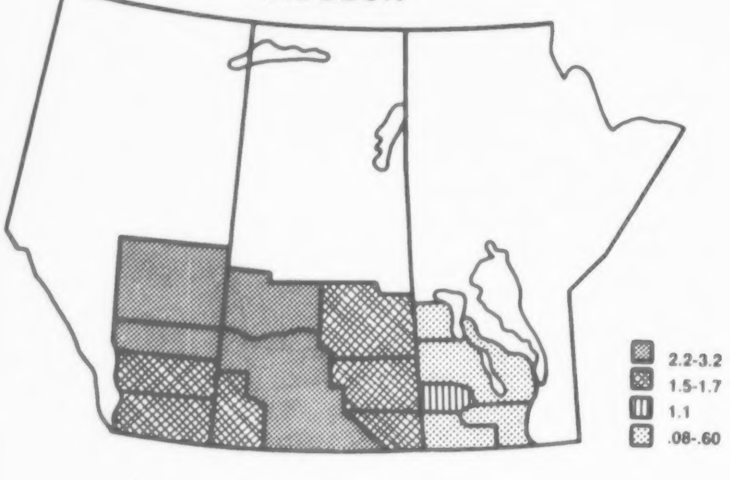
MALLARD


▨ 9.6-12
▩ 7.8-8.6
▥ 4.2-6.7
░ 1.1-4.0

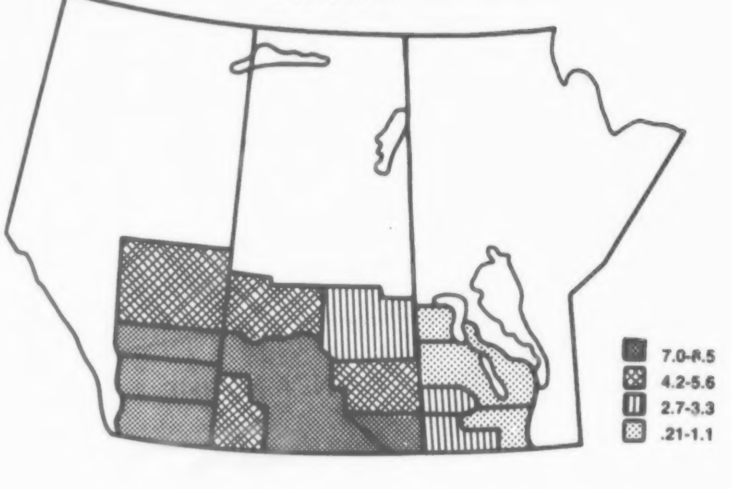
GADWALL


▨ 2.8-2.9
▩ 1.9-2.1
▥ .95-1.7
░ .07-.39

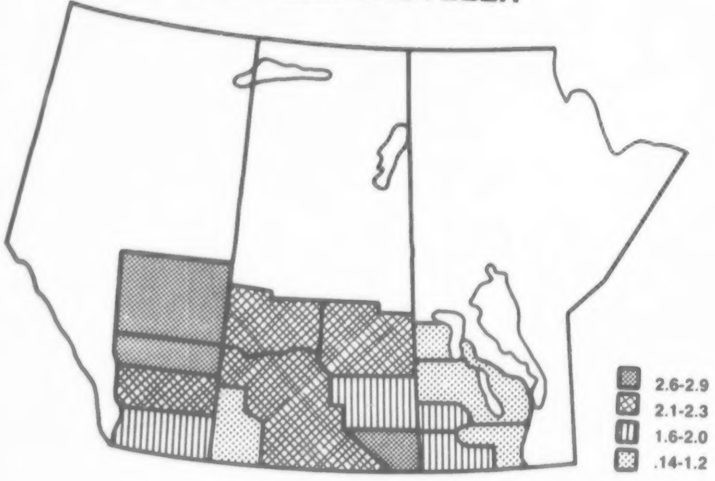
WIDGEON


▨ 2.2-3.2
▩ 1.5-1.7
▥ 1.1
░ .08-.60

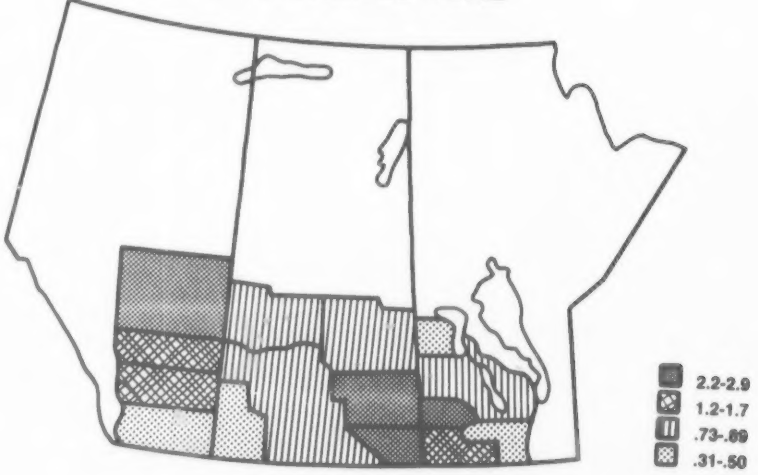
PINTAIL


▨ 7.0-8.5
▩ 4.2-5.6
▥ 2.7-3.3
░ .21-1.1

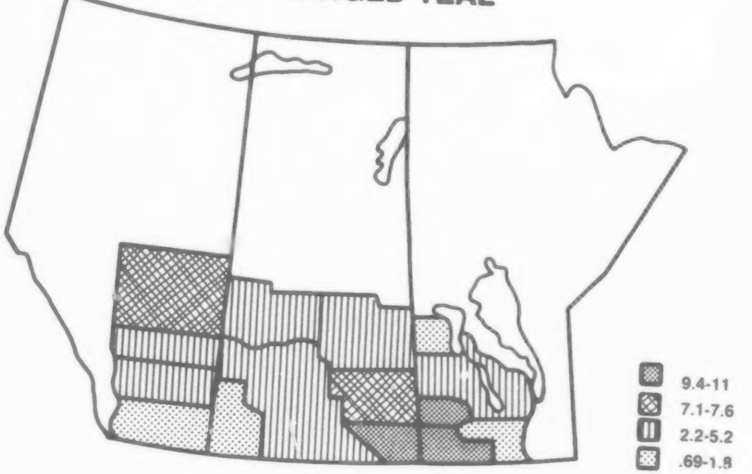
NORTHERN SHOVELER


2.6-2.9
2.1-2.3
1.6-2.0
.14-1.2

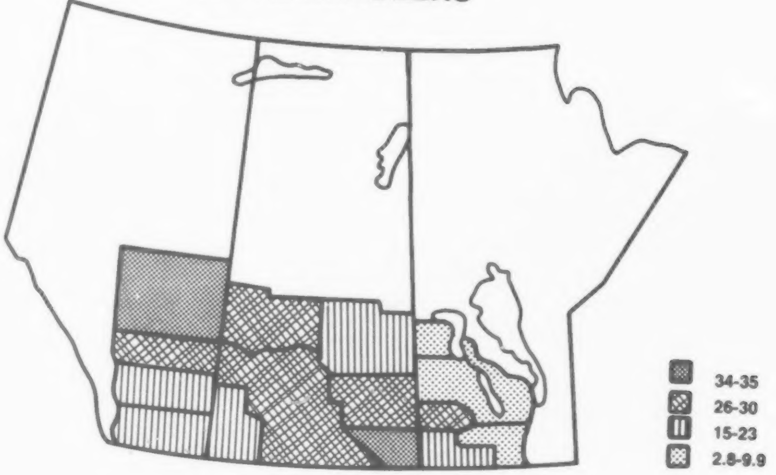
GREAN-WINGED TEAL

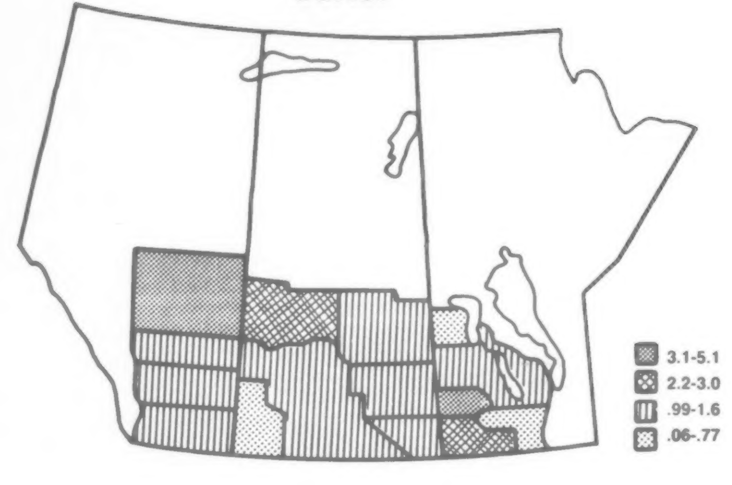

2.2-2.9
1.2-1.7
.73-.89
.31-.50

BLUE-WINGED TEAL


9.4-11
7.1-7.6
2.2-5.2
.69-1.8

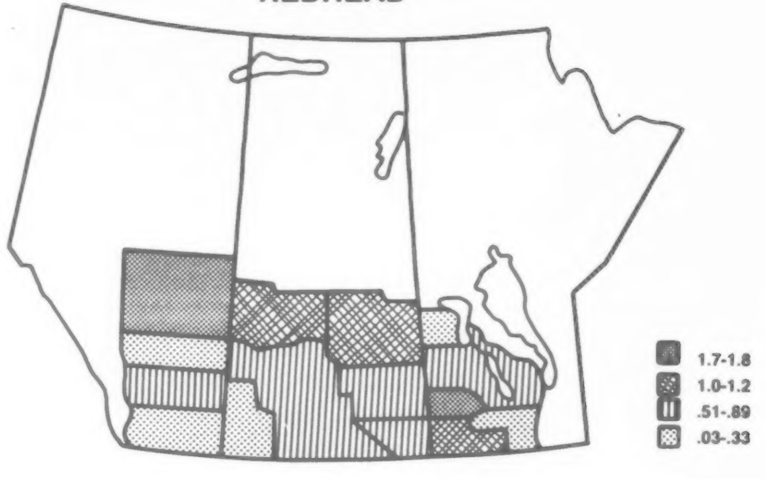
TOTAL DABBLERS


34-35
26-30
15-23
2.8-9.9



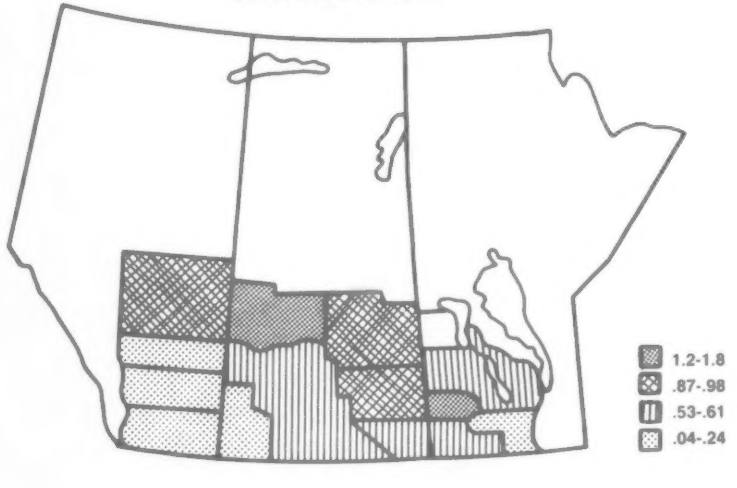
SCAUP

3.1-5.1
2.2-3.0
.99-1.6
.06-.77



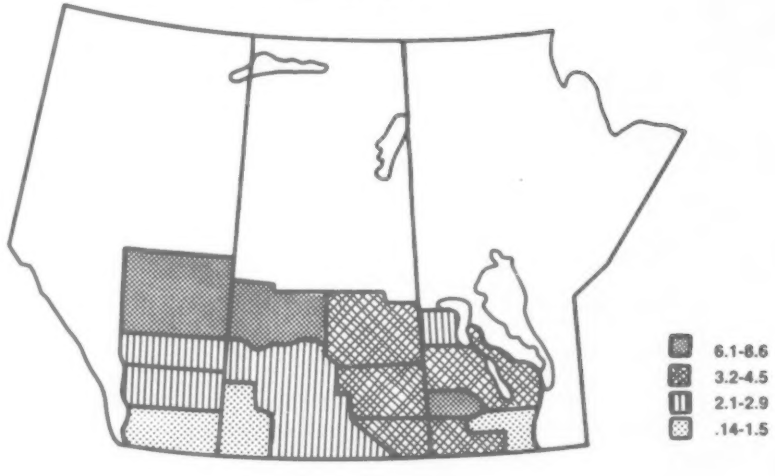
REDHEAD

1.7-1.8
1.0-1.2
.51-.89
.03-.33



CANVASBACK

1.2-1.8
.87-.98
.53-.61
.04-.24



TOTAL DIVERS

6.1-8.6
3.2-4.5
2.1-2.9
.14-1.5

drought in the southern prairies and loss of both quantity and quality of habitat as a result of agricultural practices such as pesticide use.

It will become evident that our ability to quantify the relative importance of these possible stress factors is severely limited by the quality of the existing databases. What is clear is that any factor that results in a serious negative impact on the parklands and grasslands will have serious implications for the North American duck population as a whole. However, due to differences in distribution, harvest rates and habitat utilization between various species, changes that affect one region or habitat type should not be expected to have the same consequences for all species.

3.2 Quantitative Analysis of Waterfowl Habitat in the Prairies

Given the number of basins, which is estimated at between six to ten million excluding sheet water (Gollop 1965), and the temporary nature of the smaller potholes, keeping a habitat inventory of prairie wetlands is a formidable task. In fact, there has never been a complete enumeration of all the potholes in the region. Two main types of surveys are available: national surveys at a scale of 1:50,000 with a resolving power of approximately 2.6 ha (McClellan *et al*. 1968) and small surveys at the 1:20,000 or 1:15,000 scale which enable the identification of potholes smaller than or equal to 0.4 ha in size but deal with a limited region. As detailed in Chapter 2, these *small potholes can be of vital importance to waterfowl production and therefore an inventory which does not incorporate them is of greatly reduced value to waterfowl managers*.

Apart from resolution problems and lack of comprehensiveness, another limiting factor with inventories is the infrequency of observation. If the quantity or quality of the wetland varies over time, as is the case with waterfowl habitat, then the value of an inventory diminishes with the lapse of time between collection, analysis and eventual utilization. Most, but not all, of the prairie wetland inventories are simply "snapshots" which provide information as to the state of the habitat when the data base was established but which may reveal very little about the condition or even the existence of the same habitat today.

In his review of wetland inventories in western Canada, Gollop (1965) listed the prairie wetland data bases which are potentially relevant to waterfowl concerns: early efforts by the USFWS (1954-57), Ducks Unlimited (DU) and the National Wildlife Federation. Since Gollop's review, some of these data bases have been expanded. For example, DU has enlarged its mapped inventory of wetlands to include potholes greater than or equal to 10 ha and has tabular data covering much of the prairies for potholes greater than or equal to 2 ha (T. Neraasen, pers. comm.). Unfortunately, all of these data bases have some limitations with respect to the information needs of this project. In addition to these efforts, a national land use inventory (Canada Land Inventory) has been developed and this will serve as the foundation for our analysis. A number of other less comprehensive wetland data sets have also been compiled, primarily to document wetland loss in various regions (see review by Lynch-Stewart 1984 updated by Rubec & Rump 1984), and these will be used to supplement data from the Canada Land Inventory.

3.2.1 The Canada Land Inventory (CLI)

In 1963, the federal government began what was eventually to become the Canada Land Inventory (CLI), a planning tool meant to serve a number of resource management interests (Lands Directorate 1978a). Working from base maps at a 1:50,000 scale, the CLI identified potential land use for most of the inhabited portion of southern Canada based on parameters such as soil type, topography, climate, flora, etc. Most of these data are available in map form at three scales 1:1,000,000, 1:250,000 and 1:50,000 for agricultural, waterfowl, ungulate, forestry and recreational potential. These data sets have been digitized at the 1:250,000 scale and are available on a computer data base. Actual (circa 1967) land use is available on data base only. With the exception of the current land use information, which was compiled from a human perspective (i.e., urban, agricultural, woodland, wetland or unproductive land), the CLI data represent the potential for a given resource (such as waterfowl) and not actual use. However, with the aid of additional information such as agricultural or waterfowl production surveys and the CLI current land use data, it is possible to make some judgements as to the actual use of the land by these interests. The computerized CLI is the only data base that includes the entire parkland and grassland region for both agriculture and waterfowl production potential, the primary foci of this report. The following can serve as an introduction to both.

The CLI generally recognizes eight primary classes of resource potential. Class 1 represents areas that have no significant limitations to a particular resource, whereas Class 8 areas have such severe limitations that little or no potential exists for that resource. Within the primary waterfowl Classes 1 through 3, there are special designations for habitat that also serves as important migration stops or wintering areas (see Lands Directorate 1973 for details of classification). With the exception of Class 1, subclasses can also be used to identify significant limiting factors for a particular resource. Since this level of detail is beyond the needs of our analysis, only the primary classification for both the waterfowl and agriculture potential are utilized.

CLI data at the 1:250,000 scale cannot resolve individual features on the landscape (i.e. potholes) but rather, habitat or soils of a similar classification are encompassed within a polygon of up to three primary classes depending on the map scale and topography. This "complexing" of primary classes is done in such a way that the proportion of any one class within a polygon is known but not its location. Most of the analyses presented in this chapter were performed on the basis of such "complexed" data. The exception is the regional analysis of agriculture-waterfowl overlap presented in Section 3.4.2. This analysis was performed on de-complexed data which presumably is more accurate since all distinct classes are tallied for each polygon.

Another important aspect of the CLI data base is that the data for the original base maps were collected between 1965 and 1971. As will become apparent in Section 3.3, agricultural pressure in the prairies since 1965 may have reduced the waterfowl productivity but enhanced the agricultural capability in many areas thus changing both the quantity and classification of the affected habitat.

3.2.2 The CLI waterfowl data

The waterfowl potential of lands assessed under the CLI was established in a subjective fashion by experienced waterfowl biologists. This method may be much more accurate than any stringent criterion based on wetland density or size since it incorporates many hard-to-quantify features such as wetland diversity and edge development. Nevertheless, the presence of permanent or semi-permanent wetlands was one of the most important factors used in the rating (G. Adams, pers. comm.). Classes 1 and 2 were assigned to areas of high waterfowl potential. Class 3 land is considered to have slight limitations, Class 4 has moderate limitations and Classes 5 through 7 have severe limitations to waterfowl potential. Waterfowl Class 8 is used as a "catch-all" category that includes urban areas, water areas, forest reserves or parks. Patterson (1978) chose to include this category in his analysis and the data presented here reflect this choice. It is assumed that Class 8 represents habitat that is not at all suitable for waterfowl.

As expected from Section 3.1, the CLI waterfowl potential maps reveal that the vast majority of Class 1 through 3 and a good portion of Class 4 lands are situated within the Aspen Parkland Zone. The distribution of the best (Class 1-3) waterfowl lands can generally be visualized as a broad diagonal strip which, from west to east, begins at Edmonton and runs southeast through to the southwestern corner of Manitoba.

From the point of view of quantifying waterfowl habitat a difficulty arises in that the CLI extends beyond the grassland-parkland region of the 3 prairie provinces. In order to confine the analysis to areas that are both important to waterfowl and agriculture, Patterson (1978) obtained a CLI analysis of a more restricted land base which roughly approximates the grassland-parkland zones as defined by Bellrose (1979) together with the addition of two more strata (36 and 37) from Manitoba. The correspondence between the full CLI land base, Patterson's restricted CLI data set and existing waterfowl strata is given in Appendix 3.5. For the purpose of analysis, Patterson's restricted CLI data set was further divided into "Waterfowl Management Districts" (Alberta, SW and SE Saskatchewan and SW Manitoba - Table 3.4). Quantitative analysis of this restricted data base revealed that a large proportion of the southern prairie provinces (49% or 273,434 km^2) is classified as Class 4-5 land or, in other words, as having moderate to severe limitations (e.g. moisture) to waterfowl production. Approximately 25% of the total area (137,357 km^2) is prime waterfowl habitat (Classes 1-3). A disproportionately large part of this prime waterfowl habitat (53% of Class 1-2 and 64% of Class 3) lies in Saskatchewan.

Table 3.4. CLI potential waterfowl habitat distribution by management district[a].

Management District	Pooled CLI Waterfowl Class	Area km^2	% of District Waterfowl Class (1-8)	% of Prairie Total Within Designated Class(es)
Alberta	1-2	22,364	13	37
	3	17,317	10	22
	4-5	51,036	31	19
	(1-5)	(90,717)	(54)	(22)
	6-8	75,814	46	53
	(1-8)	(166,531)		(30)
Southwestern Saskatchewan	1-2	11,075	6	18
	3	21,916	12	28
	4-5	106,966	61	39
	(1-5)	(139,957)	(80)	(34)
	6-8	35,711	20	25
	(1-8)	(175,668)		(32)
Southeastern Saskatchewan	1-2	21,230	19	35
	3	27,819	25	36
	4-5	57,391	51	21
	(1-5)	(106,440)	(95)	(26)
	6-8	5,690	5	4
	(1-8)	(112,130)		(20)
Manitoba	1-2	5,390	5	9
	3	10,246	10	13
	4-5	58,041	58	21
	(1-5)	(73,677)	(73)	(18)
	6-8	26,628	27	18
	(1-8)	(100,305)		(18)
Prairie	1-2	60,059		11
	3	77,298		14
	4-5	273,434		49
	(1-5)	(410,791)		(74)
	6-8	143,843		26
	(1-8)	(554,634)		

[a] Based on Patterson 1978

3.2.3 Validation of the CLI waterfowl data

Patterson's (1978) attempt to correlate CLI classes with observed waterfowl densities from the USFWS and CWS waterfowl breeding surveys is presented in Table 3.5. The estimated duck densities based on the air/ground surveys agreed with the ranking of the CLI waterfowl land classification. Class 1 and 2 lands had densities ranging from 69 to 85 ducks/km^2 across all provinces whereas Class 3 lands had between 36 to 62 ducks/km^2 and Classes 6-8 had 6 to 9 ducks/km^2.

The exercise has many shortcomings primarily beause neither data base was developed with this objective in mind. The principal difficulty, which Patterson himself recognized, is the very small number of transects available for the computation of duck densities by land class. Of the 60 air/ground transects potentially available, 25 were selected since only those transects lay completely within a single waterfowl class. To compensate for this, Patterson pooled several classes (e.g. 1-2, 4-5, 6-7-8) and then adjusted regional density estimates on the basis of surveyed duck and pond numbers averaged over a 10-year period within each of the waterfowl management units. The final outcome is a series of breeding bird density estimates by land class, or group of land classes. Only for Mallards are the data "robust" enough to withstand this degree of manipulation. Total duck numbers are calculated assuming a set proportion of Mallards to total ducks for each of the waterfowl districts. The reader is referred to Patterson (1978) for a more detailed description of the data manipulation and the numerous assumptions inherent to the analysis. Despite the shortcomings of the calculations, Patterson's work represents the only known attempt to relate classified habitat to actual waterfowl production over the entire region.

Further validation of the CLI classification and of Patterson's density estimates was provided by G. Adams (pers. comm.) for one location in S. Saskatchewan and one in Manitoba between the years 1974-77. The mean duck density/km^2 over the 3-year period on the Saskatchewan site varied from 65 to 91 ducks /km^2 for Class 1 land and from 69 to 71 ducks/km^2 for Class 3 land. On the Manitoba site, the sampled areas within Class 1 land gave mean density figures ranging from 51 to 95 ducks/km^2 and those on Class 3 land had mean densities of 31 to 56 ducks/km^2.

Additional justification for the pooling of waterfowl Classes 1 and 2 is available for Alberta transects only. This data set was compiled during the regular May waterfowl survey in 1973, using an additional person in the aircraft who notified the pilot and observer at 2-mile intervals as they flew the transects. This enabled the population counts to be assigned to specific land classes more easily than if the entire transects had been run uninterrupted. An initial analysis performed on the raw data found that Class 1 and 2 lands could not be differentiated on the basis of duck density but that there was a statistically valid difference between Classes 1 and 3 (H. Weaver, pers. comm.). Furthermore, duck densities generally declined with decreasing CLI waterfowl potential.

Table 3.5. Estimated mean duck densities for CLI waterfowl classes[a].

	Estimated Mean Duck Density (ducks/km^2)			
CLI Waterfowl Class	Alberta (strata 26,27,28,29)	SW Saskatchewan (strata 30,32,33)	SE Saskatchewan (strata 31,34,35)	Manitoba (strata 36, 37,38,39,40)
1-2	85	69	69	80
3	62	47	39	36
4-5	40	31	22	14
6-8	9	7	7	6

[a] From Patterson 1978.

When Patterson's density estimates are applied on a prairie-wide basis, total population estimates by CLI classification can be determined for the grassland and parkland area (Table 3.6). The total breeding duck population is estimated as 16.7 million for the prairies. The difference between this estimate and the 15.9 million estimated from survey data only (Table 3.1) stems from the fact that Patterson used a slightly older survey data set in his computations. Whereas Class 1-3 lands make up only 25% of the total prairie land area, they are estimated to account for 48% of the breeding duck population. The bulk of the remaining population (44%) is found in Class 4-5 lands. The latter are roughly half as productive, but account for approximately twice the surface area, as the prime Class 1-3 lands. The majority of the breeding population (53%) was found in Saskatchewan, equally divided between Class 1-3 and Class 4-5 lands. This analysis will be further broken down geographically when we consider the overlap of waterfowl habitat with agriculture in later sections of this chapter. The full significance of this analysis of waterfowl populations will become apparent when we estimate the potential of pesticides to reduce duck breeding density or breeding success (Chapter 8).

3.2.4 Current status of prairie waterfowl habitat

As mentioned above, the CLI only provides a "snapshot" of what is a very dynamic system. Given the age of the CLI data base, it is all the more important to understand historical and current processes that affect the viability of prairie wetland habitat for wildlife.

With the exception of sites specifically purchased or managed for waterfowl (estimated from a number of unpublished sources to be less than 2% of the CLI class (1-5) waterfowl habitat), changes in wetland habitat are basically in one direction only: deterioration. This is in marked contrast with agricultural lands which can be "improved" for crop or other farm uses. The usual man-induced perturbations of prairie wetlands are summarized in Table 3.7 (USNRC 1982). The major impacts are total or partial drainage of the wetlands and cultivation of the margins. Temporary wetlands are usually the most seriously impacted by drainage since they are easily drained and filled. One alternative to complete drainage is wetland consolidation in which several small wetlands are drained into a single one allowing it to flood. Large-scale consolidation programs are currently being planned for the prairie provinces. The stability of these new, artificially expanded water bodies and their reduced shoreline edge results in a lower diversity in the new species complex that inhabits them. Weller (1981b in USNRC 1982) estimated that this type of "flooding" would eliminate 65-70% of the nesting bird population as a result of the lowered habitat diversity. If this assessment is correct, consolidation should be considered as a poor substitute for wetland preservation.

The best large scale assessments of prairie wetland loss are available from the U.S. In Iowa the documented cummulative loss of wetlands since the mid-1800's stood at 94% in 1981 (Shaw & Fredine 1956 and Bishop 1981, in USNRC 1982). Losses were greatest between 1906 and 1922 (3% per

Table 3.6. Estimated mean duck numbers by CLI land class in the four prairie "duck management districts"[a].

Management District	Pooled CLI Waterfowl Class	Estimated Number of Ducks	% of District Ducks in Class 1-8	% of Prairie Ducks Within Designated Class(es)
Alberta (Str. 26,27, 28,29)	1-2	1,900,940	33	42
	3	1,073,654	19	30
	4-5	2,041,440	36	27
	(1-5)	(5,016,034)	(88)	(32)
	6-8	682,326	12	60
	(1-8)	(5,698,360)		(34)
Southwestern Saskatchewan (Str. 30,32,33)	1-2	764,175	14	17
	3	1,030,052	19	29
	4-5	3,315,946	62	45
	(1-5)	(5,110,173)	(95)	(33)
	6-8	249,977	5	22
	(1-8)	(5,360,150)		(32)
Southeastern Saskatchewan (Str. 31,34,35)	1-2	1,464,870	38	32
	3	1,084,941	28	30
	4-5	1,262,602	33	17
	(1-5)	(3,812,413)	(99)	(25)
	6-8	39,830	1	4
	(1-8)	(3,852,243)		(23)
Manitoba (Str. 36,37,38, 39,40)	1-2	431,200	24	9
	3	368,856	21	10
	4-5	812,574	46	11
	(1-5)	1,612,630)	(91)	(10)
	6-8	158,768	9	14
	(1-8)	(1,772,398)		(11)
Canadian Prairie Total (All Strata)	1-2	4,561,185		27
	3	3,557,503		21
	4-5	7,432,562		44
	(1-5)	(15,551,250)		(93)
	6-8	1,131,901		7
	(1-8)	(16,683,151)		

[a] After Patterson 1978

Table 3.7. General types of wetland losses in midcontinent North America[a].

Total loss

A. Drainage followed by farming, construction or other use of the site.

B. Tilling of adjacent areas with encroaching agricultural or other land use resulting in loss of wet meadow zone.

C. Filling from the perimeter creating abrupt shorelines and decreased size.

D. Siltation by poor uphill soil management in agricultural or urbanized areas.

Modified productivity due to pollutants

A. Fertilizer runoff
B. Livestock wastes
C. Sewage disposal
D. Pesticides

Water level modification

A. Flooding
B. Water stabilization
C. Use for irrigation, industry or human water supply

Introduction of exotic plants or animals

A. Carp and sunfish
B. Trout stocking for aquaculture
C. Aquatic plants

Cluster modification with loss of diversity of wetland types by complete or partial drainage

A. Drainage of small units into streams, reducing size of both large and small units.

B. Drainage of smaller units into larger wetlands, flooding larger wetlands.

Flooding of wetland clusters

A. Flood control impoundments
B. Irrigation impoundments
C. Hydroelectric power production

[a] From USNRC 1982

year) and since 1955 have stabilized at .2% per year. The total area of wetlands in that state used to cover approximately 445,000 ha; they now account for less than 20,000 ha. Estimates for the total U.S. prairie are that only 35% of the original 29.8 million ha of wetlands remain. At current rates of loss, most of the remaining wetlands will be lost by the year 2055.

No such overviews are available for the Canadian prairies but Gollop (1965) stated that "we have no reason to expect that the pattern will be different in Canada." Millar (1982, 1983, 1984) estimated that only .5% of basins along the annual air/ground surveys had been totally drained and/or filled. However, in view of some of the other data available (Rubec & Rump 1984), it would appear that the area included within the air/ground surveys is not representative of their respective strata in that they underestimate the number of wetlands lost to drainage and cultivation (F.G. Cooch, pers. comm.). This is because survey transects usually follow good roads. The roads can act as dams and drainage tends to be away from fields and toward the roads (G. Adams, pers. comm.). In addition, farmers are less likely to farm by roads. It should be noted that the transects were selected to survey waterfowl and not agricultural impact.

Although the drainage and destruction of wetlands may have begun earlier in the U.S., from the late 60's onward the same patterns were becoming evident in Canada (Bellrose 1976). Canadian studies on prairie wetland loss have been summarized by Lynch-Stewart (1984) and updated by Rubec and Rump (1984) (Table 3.8). Despite a lack of consistancy by the authors of these studies in their definition of a wetland, in most, if not all cases, wetlands being considered would be of the temporary type or better. It is noteworthy that the loss rates given for the Minnedosa pothole region and for the Alberta parklands are in the range of the worst loss rates documented for Iowa above. Of special concern is an updated study by Rakowski and Chabot (1983) which showed that the wetland loss in the Minnedosa area between 1974 and 1982 was as great as it was between 1964 and 1974.

The impact of the loss of wetlands on waterfowl and other wildlife is obvious. The consequences for agriculture are not as clear. It is clear that agriculturalists regard potholes as under-utilized land which offers no tangible benefit to the farmer except when "improved" to increase the farmer's quota entitlement with the Canadian Wheat Board. However, there is evidence which suggests that these basins act as ground water recharge sites (Meyboom 1963, 1967, Lissey 1968, 1971 all in Zittlau 1979). Also, there is a growing awareness that these wetlands play an important role in reducing flood damage and that, due primarily to salinity, the land reclaimed from sloughs is typically of low agricultural productivity. Given the apparent long-term trend toward dessication of prairie soil (Moulding 1983), the relationship between slough and soil moisture conditions should be of interest to the agricultural community as well as to the waterfowl biologist. The provincial wetland drainage and consolidation plans mentioned above indicate that the potential value of wetlands to agriculture is still being ignored. These points are somewhat tangential to the focus of this report (see Zittlau 1979 and Brace & Pepper 1984 for detailed discussion) but they are important nevertheless since, in order to place

Table 3.8. Quantitative wetland land use conversion studies in western Canada.

	Region and Authors	Study Area Size	Era	Net Wetland Change
1[a]	Aspen Parkland (Alta.) (Shick 1972)	699 km^2	1900-1970	61% loss (area)
2[a]	Battle River Basin (Alta.) (Ritter 1979, 1980)	301 km^2	1800-1978	9% loss (area)
3[a]	South Saskatchewan River Basin (Alta.) (Schmitt 1980)	45 km^2	1800-1979	21% loss (area)
4[a]	Southern Saskatchewan (Sask.) (Millar 1981)	82 km^2	1950-1980	27% permanent impact (no. of sites)
5[a]	Black Soil Zone (Alta., Sask., Man.) (Goodman & Pryor 1972)	21 km^2	1940-1970	13% loss (area)
6[a]	Newdale Plain (Man.) (Adams & Gentle 1978)	248 km^2	1964-1974	17% affected 7% eradicated
7[a]	Minnedosa Pothole Region (Man.) (Kiel _et al_. 1972)	131 km^2	1928-1964	27% loss (area)
8[a]	Minnedosa Pothole Region (Man.) (Rakowski _et al_. 1974)	131 km^2	1964-1974	40% loss (area)
9[b]	Valley River Watershed (Man.) (Pokrant & Gaboury 1983)	1,786 km^2	1948-1981	37% loss
10[b]	Minnedosa Pothole Region (Man.) (Rakowski & Chabot 1983)	131 km^2	1974-1982	41% loss (area)
11	Odanah (Man.) (Zittlau 1979)	88 km^2	1968-1978	15% loss (area)
12	Strathcona (Man.) (Zittlau 1979)	128 km^2	1968-1978	3% loss (area)

[a] References in Lynch-Stewart 1984

[b] References in Rubec and Rump 1984

pest control strategies into proper context, we must have an understanding of the waterfowl resource and of the factors affecting it. Given a declining habitat base, any demonstrated impact on the remaining habitat acquires a greater relative importance. Also, it is clear that there is a very poor understanding of even the most basic principles relating to prairie ecology and that the destruction of the wetland resource in the face of this ignorance is symptomatic of the mismanagement of a region that is vital to both agriculture and waterfowl.

The second aspect of wetland health which was mentioned earlier is margin development. A wetland which has been cleared of its vegetation border or which is tilled right up to the edge is of no use to waterfowl for nesting. Merriam (1975, 1978) studied 913 sloughs throughout Alberta's parkland and compared 398 with aerial photographs taken between 1945-1949. Between 1945-49 and 1974, 32% of the sloughs had had their borders completely removed and an additional 48% had reduced borders as a result of clearing for agricultural use. In the annual air/ground surveys between 1980 and 1984, Brace and Caldwell (1985) documented that approximately 50% of slough margins (arbitrarily defined as 10 m from the high water mark) sampled in southern Saskatchewan and southwestern Manitoba have had their margins physically transgressed. This proportion rose to almost 90% in southern Alberta. As with the practice of draining sloughs, it is not at all clear that the practice of burning or clearing slough margins produces any substantial economic benefit to agriculture (Merriam 1978). In Merriam's slough sample, agricultural activity was taking place within 50 m of 94% of sloughs in the southern parklands and 30% of sloughs in the rest of the Alberta parklands. This latter figure is probably much greater today with the increased push to putting marginal lands into production. It is also worth restating that the loss of the margin cover reduces snow entrapment and contributes to surface water evaporation (see Section 2) which has obvious implications for the farming community. Clearing slough borders has the added effect of facilitating access by heavy machinery which allows cultivation closer to the slough edge and greater pesticide input into the slough proper. These rates of margin loss do not take into account margin degradation as a result of herbicide use. This factor has largely been ignored by waterfowl biologists but, as we shall demonstrate in Chapters 7 and 8, there is cause for concern.

Intensive efforts to quantify habitat loss and degradation on a larger scale are ongoing. For example, the air/ground transects were photographed over a three year period during the May and July duck surveys of the strata so as to provide an accurate basis for measuring future changes (R.K. Brace, pers. comm.). Satellite imagery has also been used in an attempt to provide biologists, at minimal cost, with an assessment of wetlands over a larger area. However, the current technology cannot resolve the small but important sloughs and recent attempts to use this technology in Saskatchewan have failed (J.B. Millar, pers. comm.). Even larger wetlands may not be discernable by remote sensing in years of drought.

3.3 Agricultural Land Use in the Prairies

3.3.1 CLI agriculture potential and actual agricultural use

As discussed in Section 3.2, the capability of lands for agriculture is assigned in a manner analogous to the designation of the potential for waterfowl but on the basis of soil type. Class 1 soils have no significant limitations to use, Class 2 and 3 have moderate limitations and Class 4 soils have severe limitations that restrict the range of crops, require special conservation practices or both. Although Class 5-7 lands are occasionally put under some form of cultivation (usually pasture), the first 4 classes represent, for all intents and purposes, the sum total of arable lands. However, other limitations (e.g. number of growing days) not considered in the classification scheme may also affect the potential. The complexing of agricultural capability classes tends to overestimate the amount of arable land by "hiding" the small proportions of lower capability classes, much of which could be valuable wetland (G. Adams, pers. comm.). This should not affect our hazard evaluation since the overlap estimates were computed from decomplexed data (Section 3.4.2).

As with the waterfowl data (Section 3.2), the bulk of our analysis will be performed on a subset of lands classified by the CLI which corresponds to the grassland-parkland region of the prairies as defined by Patterson (1978). This region encompasses most of the valuable agricultural land (Appendix 3.6). Once again, the main omission is the Peace district in Alberta. The provincial breakdown of the total land area under CLI Classes 1 to 7 for the grassland-parkland area is presented in Appendix 3.6.

These totals, however, only reflect potentials and not actual cropping practices. One measure of how agricultural potential is realized is available from the "present land use" compilation of the CLI (Lands Directorate 1976) which includes all lands classified under the CLI. The proportions of each land class under cultivation at the time of compilation are given in Table 3.9. (The full data are presented in Appendix 3.7.) The figures include croplands left fallow. It is noteworthy that Alberta had the least proportion of cultivated 1-3 land and Saskatchewan showed extensive cultivation of marginal (Class 4) and very marginal (Class 5 and higher) agricultural lands. Unfortunately, this compilation dates back to 1967 and is now of limited value.

A more up to date analysis of agricultural practices in the prairie provinces, which also provides for a trend analysis, is available in the form of the quinquennial agricultural censuses by Statistics Canada. Data from every second census over the last 40 years (1951-1981) for the prairie provinces as a whole as well as for each province separately are presented in Tables 3.10 to 3.13. Of note is the large increase of land under crop in all three provinces (34% in 30 years) as well as an increase in pasture area (129% in 30 years). The latter increase is especially evident in Alberta (251% in 30 years). These intensifications of

Table 3.9. Proportion of Agricultural Class 1-7 lands in cropland[a], by province.

	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7
Manitoba	94%	87%	70%	31%	23%	13%	4%
Saskatchewan	87%	76%	71%	54%	41%	14%	19%
Alberta	88%	77%	57%	26%	8%	14%	4%
All provinces	89%	80%	67%	33%	25%	13%	4%

[a] Includes pasture and forage crops (other "improved lands") as well as summer fallow

Table 3.10. Agricultural land-use change in the prairies (in thousands of ha[a]) between 1951 and 1981[b].

	1951	1961	1971	1981	Total Change	% Change
Total land area	175,604	175,604	175,604	175,604	0	0
Total number of farms	248,716	210,442	174,653	154,816	93,900	- 37.7
Total area of farms	50,121	52,533	54,053	52,671	2,550	+ 5.09
Improved land	29,071	32,523	35,487	37,712	8,641	+ 29.7
Under crops	18,400	19,112	22,078	24,602	6,202	+ 33.7
Pasture	1,270	1,531	2,198	2,908	1,638	+129
Summer fallow	8,728	11,274	10,611	9,507	778	+ 8.91
Other improved land	673	607	598	693	20	+ 2.97
Unimproved land	21,048	20,007	18,567	14,958	6,090	- 28.9
Woodland	3,085	2,356	1,467	1,022	2,063	- 66.9
Other unimproved land	17,965	17,656	17,099	13,936	4,029	- 22.4

[a] Except total number of farms

[b] Source: Census of Canada 1981, Agriculture Catalogue 96-903.

Table 3.11. Agricultural land-use change in Alberta (in thousands of ha[a]) between 1951 and 1981[b].

	1951	1961	1971	1981	Total Change	% Change
Total land area	63,823	63,823	63,823	63,823	0	0
Total number of farms	84,315	73,212	62,702	58,056	26,259	- 31.1
Total area of farms	17,992	19,112	20,034	19,108	1,116	+ 6.20
Improved land	9,012	10,233	11,517	12,525	3,513	+ 39.0
Under crops	5,839	6,319	7,322	8,441	2,602	+ 44.6
Pasture	450	676	1,110	1,581	1,131	+251
Summer fallow	2,507	3,015	2,836	2,205	302	- 12.0
Other improved land	217	2,211	248	297	80	+ 36.8
Unimproved land	8,979	8,878	8,517	6,583	2,295	- 25.6
Woodland	1,160	865	674	493	667	- 57.5
Other unimproved land	7,820	8,013	7,842	6,090	1,730	- 22.1

[a] Except total number of farms

[b] Source: Census of Canada 1981, Agriculture Catalogue 96-908

Table 3.12. Agricultural land-use change in Saskatchewan (in thousands of ha[a]) between 1951 and 1981[b].

	1951	1961	1971	1981	Total Change	% Change
Total land area	57,011	57,011	57,011	57,011	0	0
Total number of farms	112,018	93,924	76,970	67,318	44,700	-39.9
Total area of farms	24,954	26,068	26,327	25,947	993	+ 3.98
Improved land	15,704	17,449	18,788	19,683	3,979	+25.3
Under crops	9,593	9,681	11,064	11,741	2,148	+22.4
Pasture	583	564	792	975	392	+67.2
Summer fallow	5,202	6,952	6,701	6,704	1,502	+29.9
Other improved land	326	251	230	263	63	-10.3
Unimproved land	9,249	8,618	7,539	6,263	2,986	-32.3
Woodland	1,192	888	404	281	911	-76.4
Other unimproved land	8,058	7,735	7,135	5,982	2,076	-25.7

[a] Except total number of farms
[b] Source: Census of Canada 1981, Agriculture Catalogue 96-908

Table 3.13. Agricultural land-use change in Manitoba (in thousands of ha[a]) between 1951 and 1981[b].

	1951	1961	1971	1981	Total Change	% Change
Total land area	54,770	54,770	54,770	54,770	0	0
Total number of farms	52,383	43,306	34,981	29,442	22,941	-43.8
Total area of farms	7,175	7,353	7,692	7,616	441	+ 6.15
Improved land	4,355	4,841	5,182	5,504	1,149	+26.4
Under crops	2,968	3,112	3,692	4,420	1,452	+48.8
Pasture	237	291	296	352	115	+48.5
Summer fallow	1,019	1,307	1,074	598	421	-41.3
Other improved land	130	132	120	133	3	+ 2.3
Unimproved land	2,820	2,511	2,511	2,112	708	-25.1
Woodland	733	603	389	248	485	-66.2
Other unimproved land	2,087	1,908	2,122	1,864	223	-10.7

[a] Except total number of farms
[b] Source: Census of Canada 1981, Agriculture Catalogue 96-908

agricultural activity seem to have taken place at the expense of woodland (e.g. Aspen parkland). Both on an absolute and a relative (%) level, the loss of woodland has been the greatest in Saskatchewan (-76% in 30 years), which can be directly attributed to the relatively high proportion of wooded areas in that province occupying Class 1-2 agricultural lands (Appendix 3.7). Saskatchewan is also the province that has been the slowest to decrease its area under summer fallow. The impact of this practice on waterfowl will be discussed below. It is interesting that the prairies (and B.C. to a lesser extent) are the only current "growth" areas in the Canadian agricultural sector (Simpson-Lewis et al. 1979, McCuaig & Manning 1982).

A comparison of the 1971 and 1981 censuses suggests that the trend to the conversion of woodlands and unimproved lands into agricultural land is not abating and, in fact, may be accelerating. According to the Canada Grains Council, the Canadian Wheat Board plans to increase Canadian grain production from an annual average of 35.6 million tonnes between 1977 and 1981 to 50 million tonnes by 1990. It is estimated that 21.3% of this anticipated increase will come from new lands being placed in production. The bulk of those lands are marginal areas (Class 4) and most of them lie in the parklands. One exception is the still partly untapped agricultural potential of the Peace River area of Alberta (McCuaig & Manning 1982). The remainder of the increased production to meet the 1990 target will come from improved yields (30.2%) and a reduction in summer fallow (48.5%) (Weaver et al. 1982).

3.3.2 Crop type and distribution

Detailed crop information from the last Agriculture census (1981) is summarized in Table 3.14. Saskatchewan has by far the least diversified agriculture with a full 67% of its cropland planted to wheat. This lack of diversity has a bearing on this report since a heavy reliance on monoculture is known to exacerbate pest problems (Ehrlich et al. 1977). Alberta has a strong beef industry and this is reflected in a strong emphasis on forage crops. Overall, forage crops make up 13% of the area under crop in the prairie provinces. This compares with 8.4% for oil seeds and 78% for cereals.

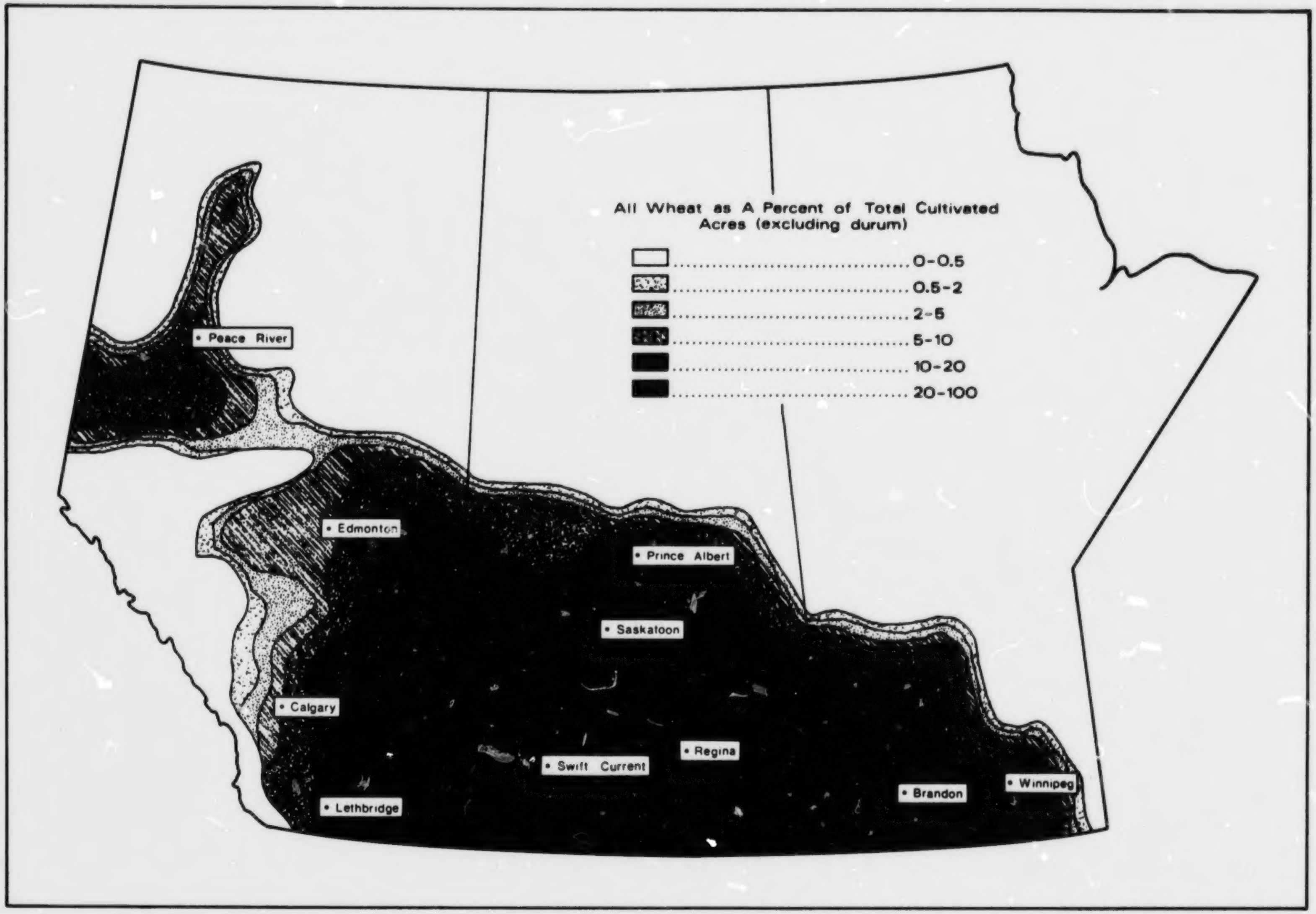
The distribution of the major prairie crops is presented in Figure 3.3. Wheat production is centered in the Saskatchewan grasslands although its distribution is widespread throughout the prairies. Oats have several foci of distribution: SW Manitoba as well as the southern parklands of Alberta and Saskatchewan. Barley and canola (rapeseed) show the most northern distribution with most of the crop produced in the parklands of all three provinces. Flax and rye have localized foci in SW Manitoba and the Alberta Parklands, respectively. Forage crops and summer fallow are widely distributed with a preponderance of the former in Alberta. A finer geographical breakdown of crop type is available from the quinquennial Agricultural censuses. This data set will be exploited more fully in coming sections linking crop practices to waterfowl numbers.

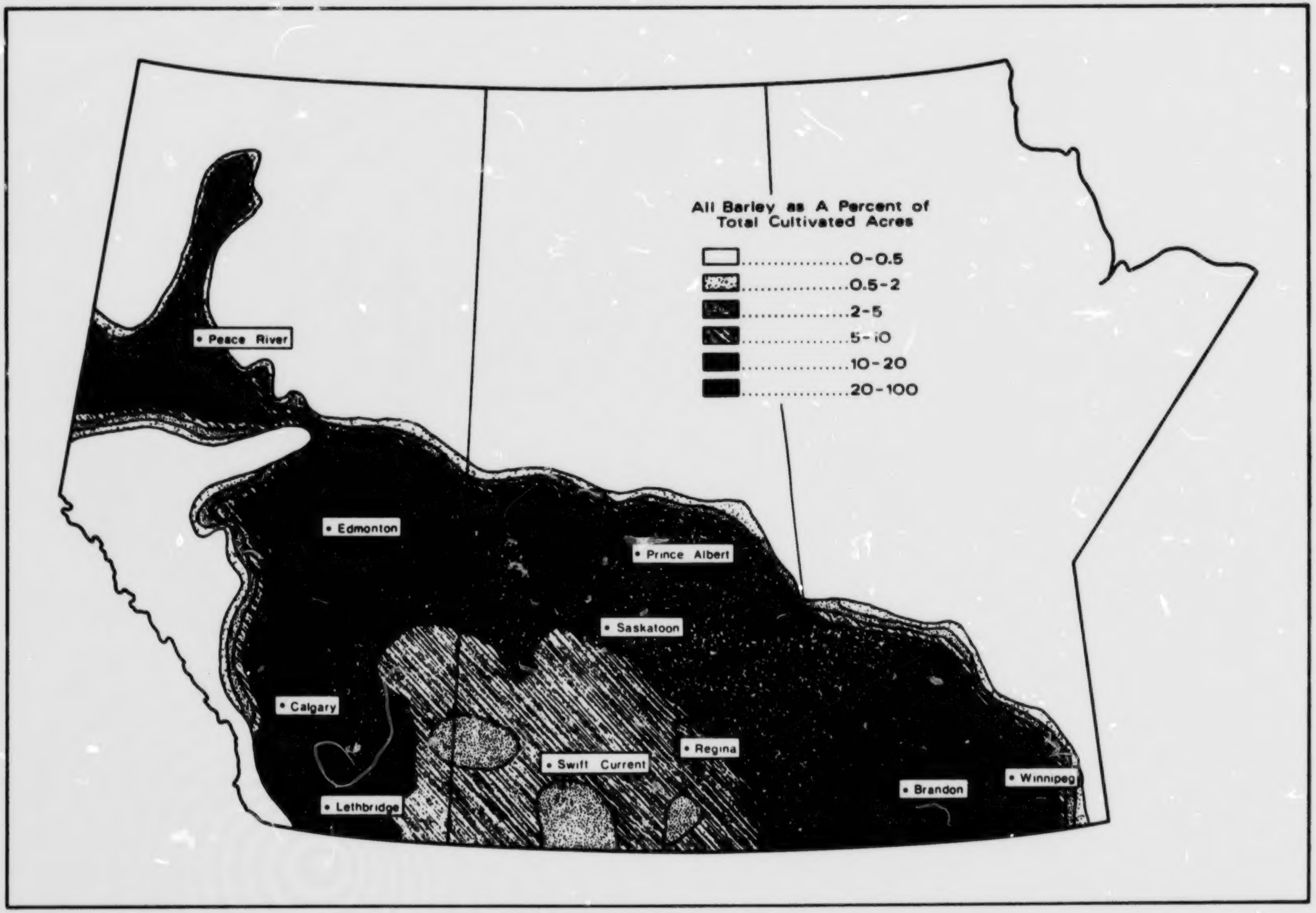
Table 3.14. Area (ha) of crops on farms, 1981[a].

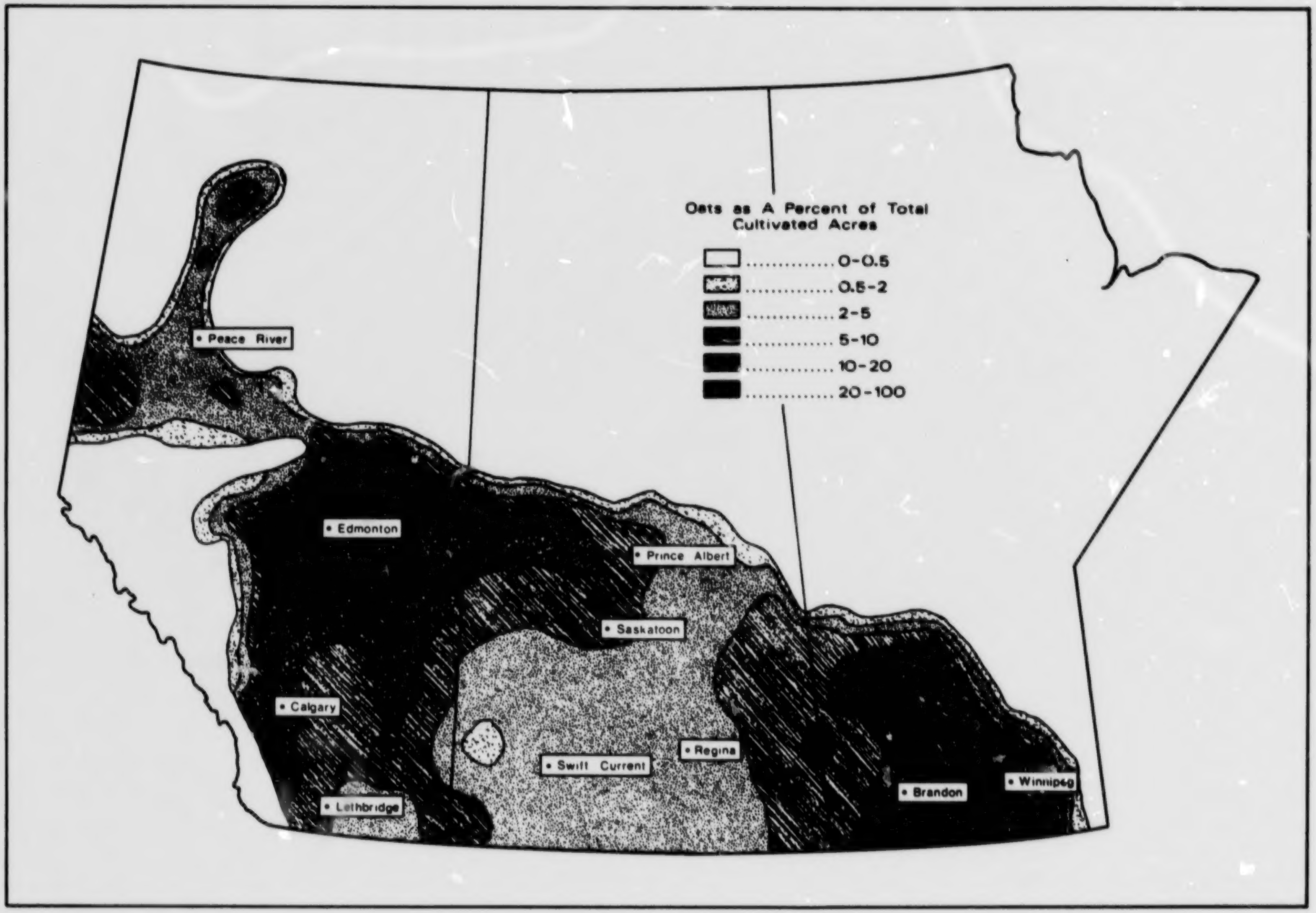
	Alberta	% of crop	Saskatchewan	% of crop	Manitoba	% of crop	Prairies	% of crop
Cereals								
Wheat	2,713,593	32.2	7,838.750	67.1	1,592,872	36.4	12,145,215	49.6
Oats	530,576	6.3	444,166	3.8	240,617	5.5	1,215,358	5.0
Barley	2,603,325	30.8	1,494,670	12.8	941,612	21.5	5,037,607	20.6
Other: Mixed grains Corn for grain Rye for grain Buckwheat	183,390	2.0	231,976	2.0	264,094	6.04	679,460	
Subtotal	6,030,884	(71.6)	10,009,562	(85.7)	3,039,195	(69.5)	19,079,641	(78.0)
Oil seeds								
Flax seed	41,418	0.5	145,286	1.2	287,470	6.5	474,174	1.9
Canola	588,901	7.0	546,712	4.7	243,009	5.5	1,378,622	5.6
Other: Soybeans Sunflowers Mustard seed	17,903	0.2	59,321	0.5	130,520	3.0	207,744	0.8
Subtotal	648,222	(7.7)	751,319	(6.4)	660,999	(15.1)	2,060,540	(8.4)
Forage								
Farm hay (largely alfalfa and alfalfa mixtures)	1,408,577	16.7	706,568	6.0	508,912	11.6	2,624,057	10.7
Other: Oats for fodder Corn for silage Other fodder crops	302,869	3.6	192,095	1.6	86,630	1.9	581,594	2.4
Subtotal	1,711,446	(20.3)	898,663	(7.7)	595,542	(13.6)	3,205,651	(13.1)
Other crops								
Includes specialty crops, vegetables, small fruits, tree fruits and nursery products	33,587	(0.4)	18,869	(0.2)	79,982	(1.8)	132,438	(0.5)
Total	8,424,139		11,678,413		4,375,718		24,478,270	

[a] Source: Statistics Canada 1981 Census of Canada - Agriculture Catalogue 96-908, 96-909, 96-910

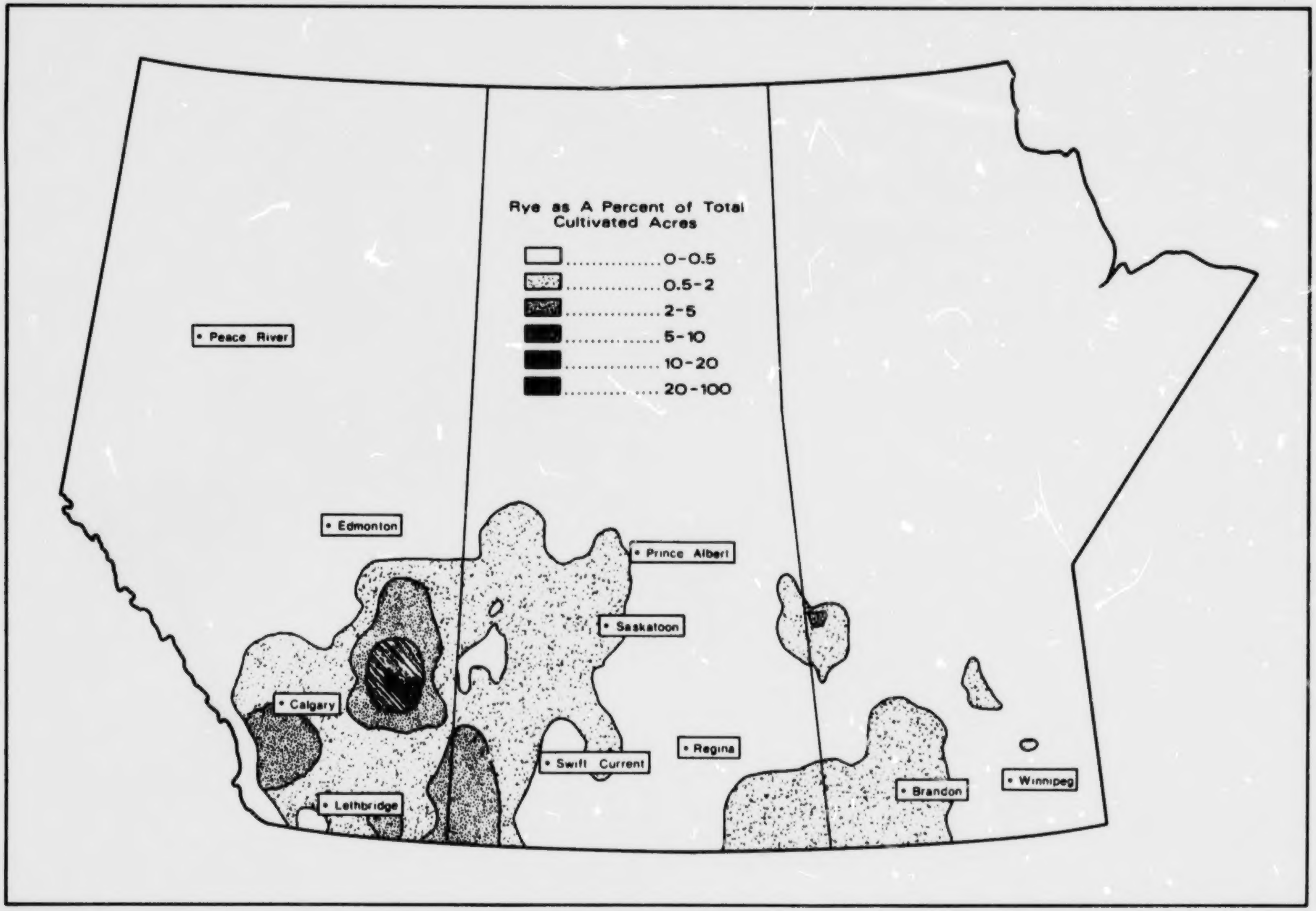
Figure 3.3. Distribution of the main crops grown in the prairie provinces in 1972 (from Alberta Agriculture 1972).

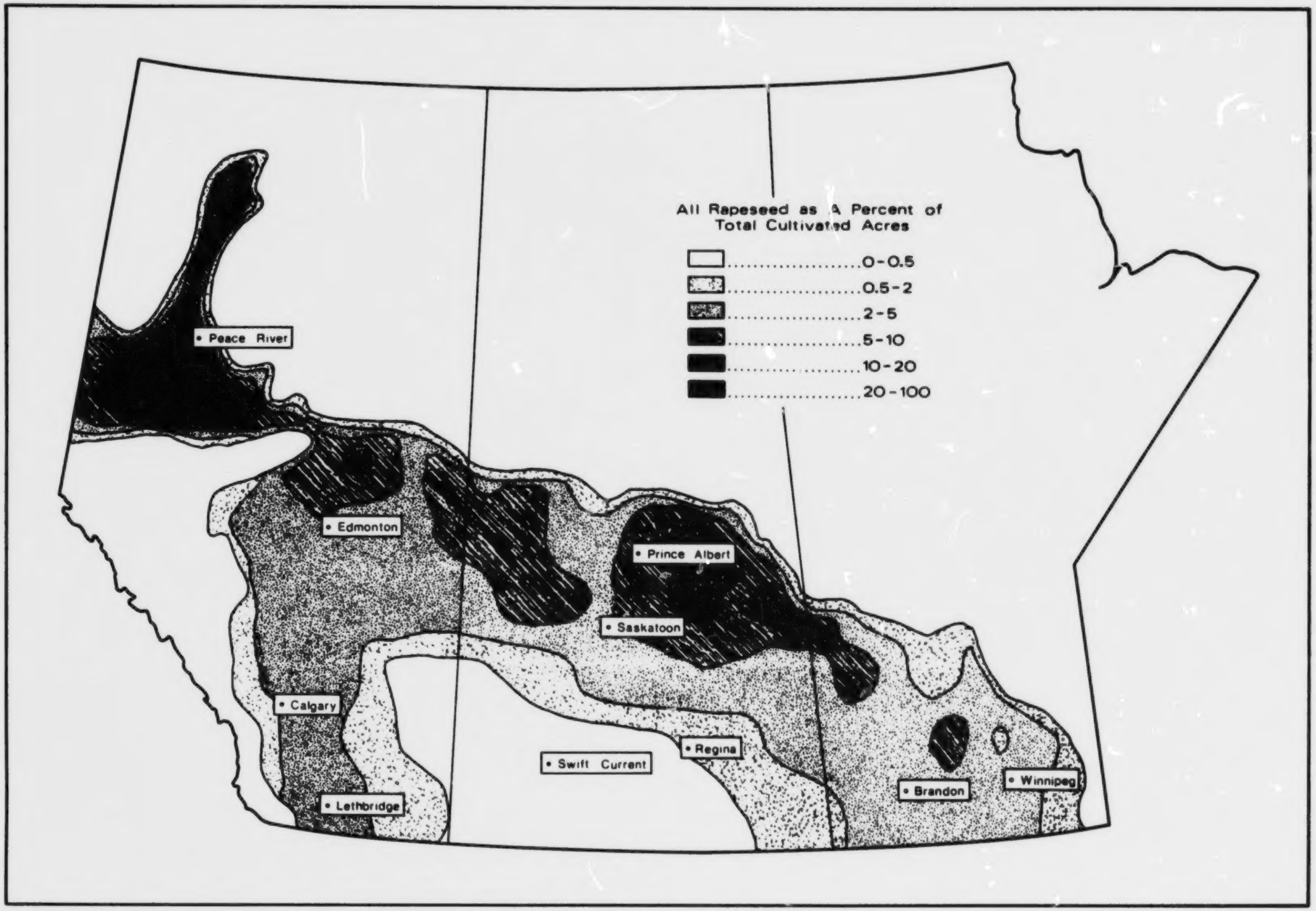

All Wheat as A Percent of Total Cultivated Acres (excluding durum)
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Prince Albert
Saskatoon
Calgary
Swift Current
Regina
Lethbridge
Brandon
Winnipeg

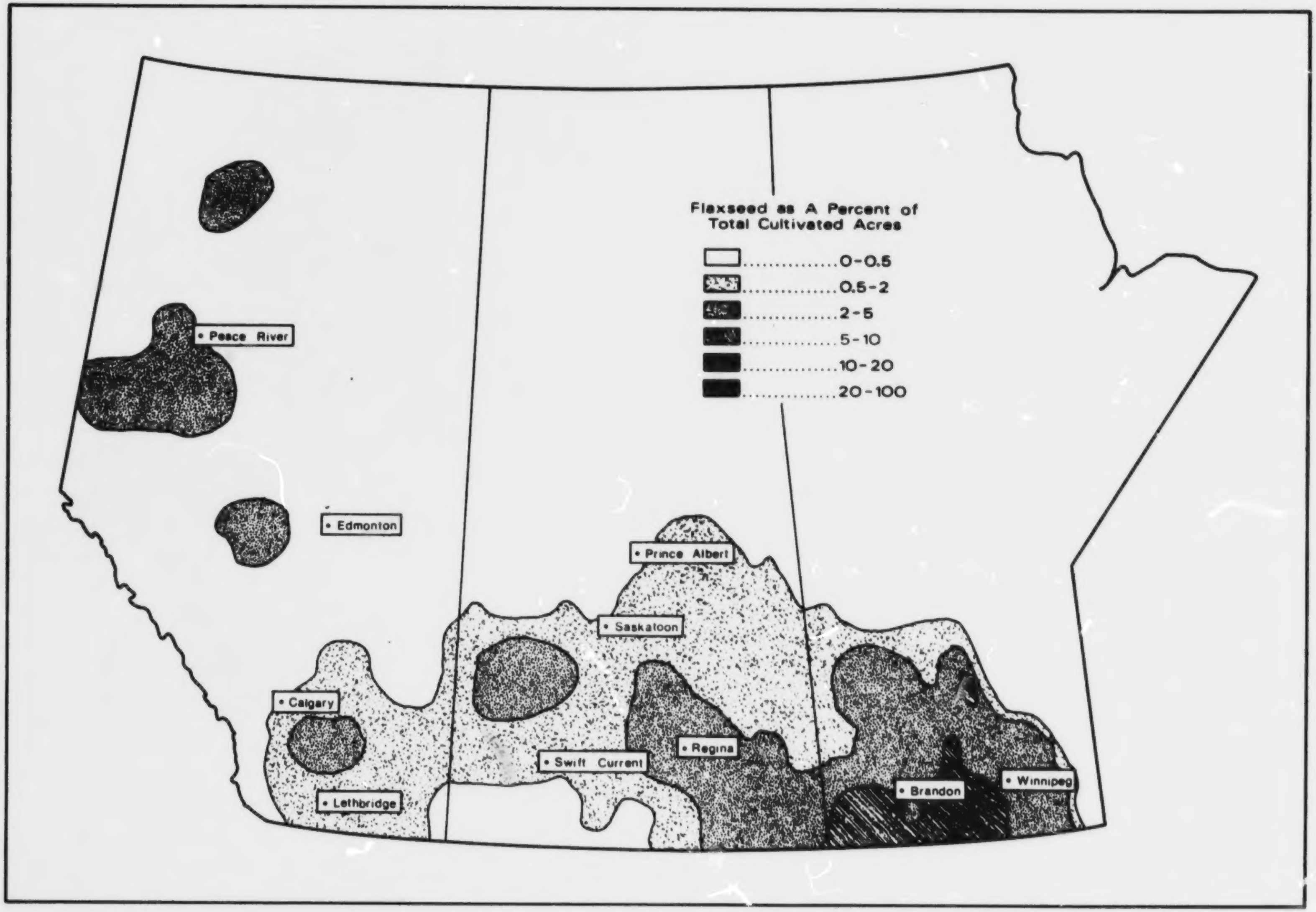

All Barley as A Percent of Total Cultivated Acres
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Prince Albert
Saskatoon
Calgary
Regina
Swift Current
Lethbridge
Brandon
Winnipeg

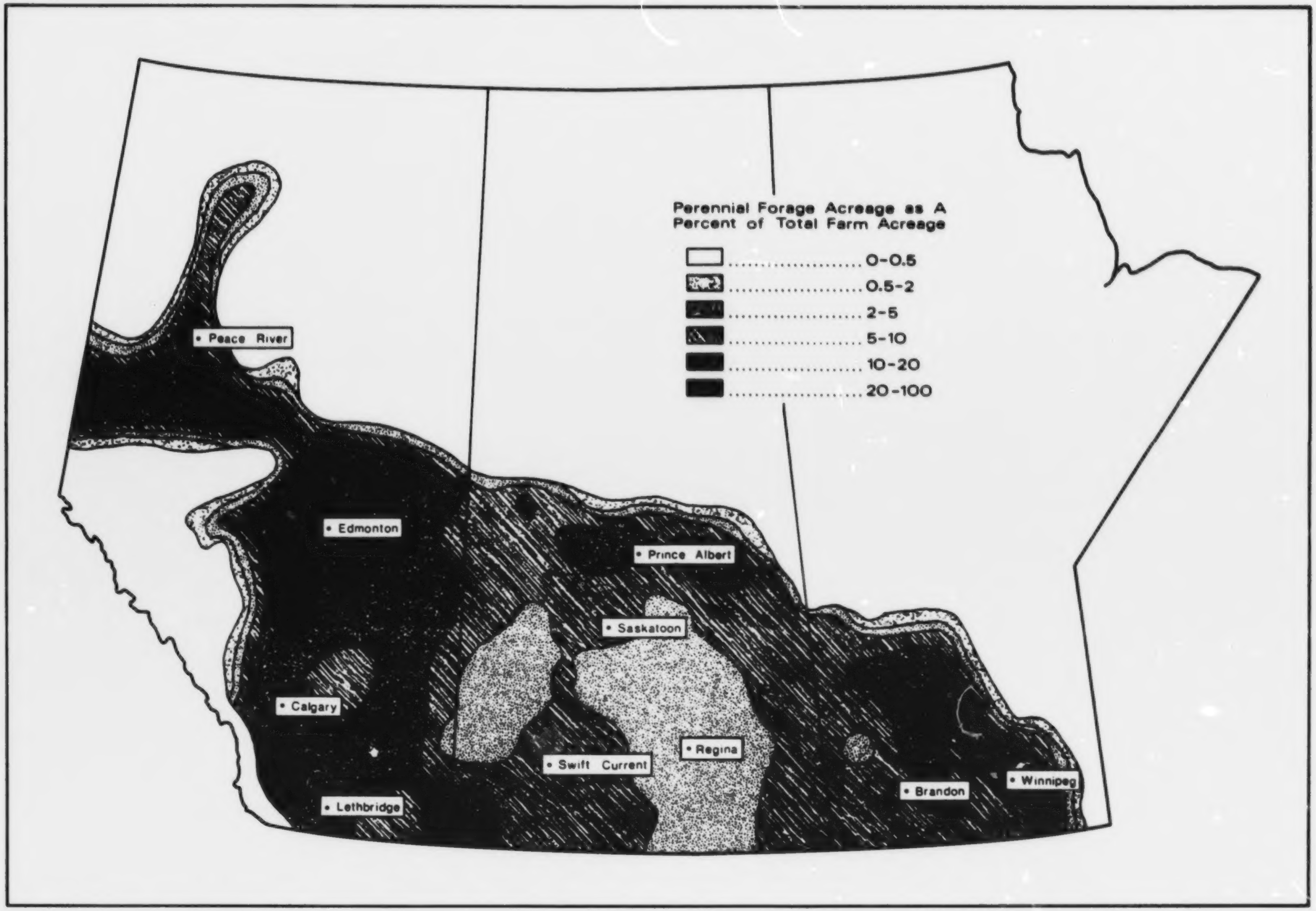


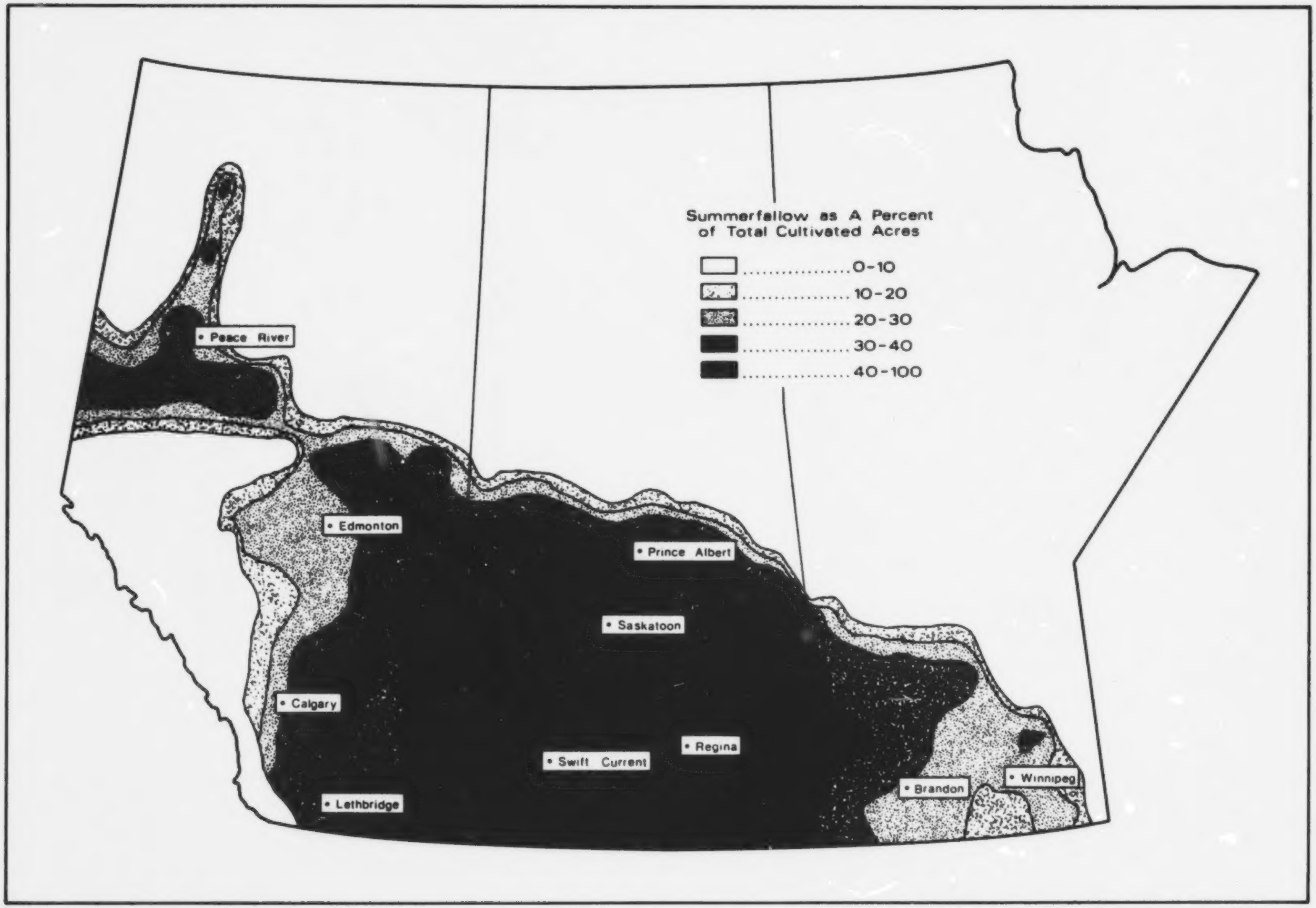
Oats as A Percent of Total Cultivated Acres
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Calgary
Lethbridge
Prince Albert
Saskatoon
Swift Current
Regina
Brandon
Winnipeg


Rye as A Percent of Total Cultivated Acres
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Calgary
Lethbridge
Prince Albert
Saskatoon
Swift Current
Regina
Brandon
Winnipeg


All Rapeseed as A Percent of Total Cultivated Acres
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Prince Albert
Saskatoon
Calgary
Regina
Swift Current
Lethbridge
Brandon
Winnipeg


Flaxseed as A Percent of Total Cultivated Acres
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Calgary
Lethbridge
Saskatoon
Prince Albert
Swift Current
Regina
Brandon
Winnipeg


Perennial Forage Acreage as A Percent of Total Farm Acreage
0-0.5
0.5-2
2-5
5-10
10-20
20-100
Peace River
Edmonton
Prince Albert
Saskatoon
Calgary
Regina
Swift Current
Lethbridge
Brandon
Winnipeg


Summerfallow as A Percent of Total Cultivated Acres
0-10
10-20
20-30
30-40
40-100
Peace River
Edmonton
Prince Albert
Saskatoon
Calgary
Regina
Swift Current
Lethbridge
Brandon
Winnipeg

According to the Canada Grains Council (Weaver et al. 1982), wheat is expected to remain the dominant crop in the prairie provinces at least until 1990 (50% of all cropland) but not to increase its dominance over other crops. Barley is expected to partly replace oats, flax seed is expected to decrease marginally and canola to increase slightly, although this market has more or less stabilized after undergoing some recent increases.

Several factors other than those of price and demand are likely to dictate future trends of crop production in the prairies. For example, summer fallowing is now recognized to have long-term negative impacts in the form of increased salinization, loss of organic matter and soil erosion (Bentley et al. 1983). Temporal diversification (rotation) of crops is also seen as a necessary step to combat salinization and the loss of organic matter (Ibid). The increase in no-till or at least low-till farming in order to reduce soil erosion will also have some impact on prairie agriculture (see Section 4.3.5).

3.4 Agriculture - Waterfowl Overlap

Having documented waterfowl populations in the prairies (Sections 3.1 and 3.2) and then the agricultural potential and realized activity in the same area (Section 3.3), the next step is to assess the potential and real overlap between waterfowl breeding habitat and agriculture.

Approximately 93% of breeding ducks in the prairies are found within CLI Waterfowl Classes 1-5 (Section 3.2). Most of the area within CLI Agriculture Classes 1-3 is currently in production and most of the perceived agricultural expansion, if it hasn't taken place already, will take place in Agricultural Class 4 (Section 3.3). In fact, the amount of land in Agricultural Classes 1-3 which is not in crop but is in pasture or some other agricultural or non-agricultural use approximates the area of land in Class 4 and up which is in crop. We shall assume that, relatively speaking, pesticide use on pasturelands and other non-croplands is minimal and therefore we shall restrict our analysis to cropland. The main exception is grasshopper spraying which does take place on non-cropland and which will be treated separately. An analysis of the overlap of Waterfowl Classes 1-5 with Agricultural Classes 1-3 should provide a reasonable index of the current extent of impingement of cropping practices on duck habitat, whereas the overlap between the same Waterfowl Classes with Agricultural Classes 1-4 should reflect the expected impact of agricultural practices on waterfowl habitat in the late 1980's and early 1990's, assuming current trends continue.

Two analyses of waterfowl-agriculture overlap have been performed by the Lands Directorate using the CLI data base. The two different data sets have already been introduced in earlier sections and compared in Appendices 3.5 and 3.6. The analysis which will form the core of this section is derived from the restricted set of lands classified under CLI, which roughly corresponds to the grasslands and parklands area of the prairies. This more focused analysis permits assignment of duck breeding pair densities (Section 3.2) to the overlap areas, a prerequisite for any hazard assessment.

3.4.1 Analysis of the overall prairie overlap

The raw overlap data for the grassland-parkland region (Patterson 1978) is tabulated in Appendix 3.8. The data are divided by waterfowl management district as defined in Table 3.5. In Tables 3.15 and 3.16 these data are expressed as percent overlap figures with respect to the total land area available within any one waterfowl capability class.

On an area basis, 63% and 80% of the Waterfowl Class 1-5 lands overlap with Agriculture Classes 1-3 and 1-4, respectively, i.e. placing all Agriculture Class 4 lands into production would increase the overlap with waterfowl-producing lands by approximately 17% over the situation where only the best agricultural lands (Classes 1-3) are cropped. The impact of placing these marginal agricultural lands in production would actually be greater on the best waterfowl lands, increasing the overlap of Waterfowl Class 1 lands from 63% to 84% or by 21%.

There is a tendency for the best waterfowl land (Class 1-2) to be more intimatly associated with agricultural lands relative to poorer waterfowl lands, at least in Alberta and Manitoba. That this relationship does not appear to hold for Saskatchewan may be an artifact of this particular analysis. As documented in Section 3.3 there has been an on-going degradation of the waterfowl habitat in areas of intensive agricultural production. Any drainage which took place before the Canada Land Inventory was conducted has had the effect of reducing the overlap between prime waterfowl land and prime agricultural land by eliminating the former. This "bias" is more likely in intensively farmed areas such as southern Saskatchewan. It is likely that under more pristine conditions, the prairies would have revealed a much stronger association between their best waterfowl lands and their best agricultural potential.

Another pertinent observation is that the overlap between the best waterfowl habitat (Classes 1 and 2) and the "current" agricultural land in production (Classes 1-3) is highest in Manitoba (87%), intermediate in Saskatchewan (62-84%) and lowest in Alberta (57%). It is interesting that this cline matches the decreasing trend in pond occupancy from west to east documented by Patterson (1978). More work is needed to determine whether this is a causal relationship.

So far, we have only dealt with waterfowl potential as it overlaps with agriculture. It is possible to use duck density figures to calculate the overlap between actual breeding duck numbers and agriculture. That analysis is presented in Tables 3.17 and 3.18 for Agricultural Classes 1-3 and 1-4, respectively. The total prairie duck breeding population from which all of the percentage values are determined is the 16.7 million figure estimated by Patterson (1978) to match CLI density values (Tables 3.5 and 3.6) across the prairies. The result of this analysis is that the overall

Table 3.15. Overlap between the stated Waterfowl Classes and Agriculture Classes 1-3 by region within the prairie (grassland/parkland) expressed as a percent of the total area within the designated Waterfowl Class(es).

Waterfowl Class	Alberta	S.W. Saskatchewan	S.E. Saskatchewan	Manitoba	Grassland/ Parkland Total
1	59%	56%	78%	93%	63%
2	52	63	86	86	75
3	57	56	84	66	68
4	55	64	86	65	70
5	42	62	61	45	54
1-2	57	62	84	87	70
1-3	57	58	84	74	69
4-5	47	63	77	52	61
1-5	51	62	80	57	63

Table 3.16. Overlap between the stated Waterfowl Classes and Agriculture Classes 1-4 by region within the prairie (grassland/parkland) expressed as a percent of the total area within the designated Waterfowl Class(es).

Waterfowl Class	Alberta	S.W. Saskatchewan	S.E. Saskatchewan	Manitoba	Grassland/ Parkland Total
1	85%	70%	85%	93%	84%
2	79	76	90	88	85
3	80	65	87	79	78
4	79	80	91	84	84
5	67	81	76	74	76
1-2	84	75	89	89	84
1-3	82	68	88	82	81
4-5	72	80	86	77	79
1-5	76	78	87	78	80

Table 3.17. Number of breeding ducks (in thousands) and % of total estimated prairie population (in brackets) within designated Waterfowl Classes which overlap with Agriculture Classes 1-3.

Waterfowl	Alberta	S.W. Saskatchewan	S.E. Saskatchewan	Manitoba	Prairie Total
1-2	1,083 (6.5%)	473 (2.8%)	1,231 (7.4%)	377 (2.3%)	3,163 (18.9%)
3	607 (3.6%)	576 (3.4%)	265 (1.6%)	245 (1.5%)	1,694 (10.1%)
4-5	958 (5.7%)	2,082 (12.5%)	966 (5.8%)	427 (2.6%)	4,433 (26.5%)
1-5	2,648 (15.8%)	3,131 (18.7%)	2,462 (14.7%)	1,049 (6.3%)	9,290 (55.5%)

Table 3.18. Number of breeding ducks (in thousands) and % of total estimated prairie population (in brackets) within designated Waterfowl Classes which overlap with Agriculture Classes 1-4.

Waterfowl Class	Alberta	S.W. Saskatchewan	S.E. Saskatchewan	Manitoba	Prairie Total
1-2	1,588 (9.5%)	575 (3.5%)	1,302 (7.8%)	385 (2.3%)	3,851 (23.1%)
3	877 (5.1%)	666 (4.0%)	943 (5.7%)	290 (1.7%)	2,757 (16.5%)
4-5	1,462 (8.8%)	2,667 (16.0%)	1,083 (6.5%)	629 (3.8%)	5,840 (35.0%)
1-5	3,907 (23.4%)	3,909 (23.4%)	3,328 (19.9%)	1,305 (7.8%)	12,449 (75.0%)

prairie overlap between duck breeding populations and agriculture is not very different from the overlap expressed on an area basis. An estimated 56% of all prairie (grassland-parkland) breeding ducks (or 9.3 million) co-exist with "current" agricultural development and this figure would rise to 75% if all Agricultural Class 4 lands were placed in production.

This analysis reveals very interesting geographical patterns which were not discernable from Tables 3.15 and 3.16. The most striking observation is the small overlap in Manitoba which stems primarily from the small area available to waterfowl and the low breeding pair densities. In Manitoba, more of the waterfowl habitat is associated with large lakes which, by definition, have a low agricultural potential. All other regions have roughly similar proportions of the total prairie breeding populations which may be impacted by agriculture (15-19%).

For Alberta and SE Saskatchewan, Waterfowl Classes 1-2 account for a large proportion of the total population "at risk" through overlap. For SW Saskatchewan, most of the duck production currently "at risk" comes from Waterfowl Classes 4-5, probably as a result of the current scarcity of better waterfowl lands.

3.4.2 Regional analysis of overlap

To assess the potential impact of a given pesticide use on the waterfowl resource, it is necessary to break down the overlap information into more manageable units than waterfowl management districts. Since basic information on crop type as well as information on pesticide use (Chapter 4) is available by census division from the quinquennial agricultural surveys, we decided to achieve this level of resolution for the waterfowl information. This task is facilitated by the published analysis of CLI capability classes by census divisions (Lands Directorate 1980).

The ultimate goal is to obtain the number of ducks "associated" (or overlapping) with any one crop type within any one census division. As a first step, we noted the amount of each waterfowl land class (from Class 1-5) in each census district (Figure 3.4). These areas were then corrected by the overlap estimates of Table 3.15. For Saskatchewan, census districts had to be ascribed to one or the other waterfowl management district. These corrected areas are our best estimate of the real overlap between the waterfowl habitat and current levels of agricultural activity. To each corrected area, we then applied the class-specific duck breeding density estimate obtained by Patterson (Table 3.4). Because of the high degree of data manipulation and high uncertainty involved in setting these density estimates, it was only possible to perform this exercise for Mallards or all ducks. We chose the latter, realizing that these data are really the Mallard data to which correction factors have been applied (Section 3.2.3).

The results of this exercise are given in Table 3.19. Discrepancies between these totals and those in Table 3.17 stem from the difficulty in assigning census divisions to the grassland-parkland zone and also the difference between, on one hand, the area under Agricultural

Figure 3.4. Map of the prairie provinces showing the overlap between census divisions and waterfowl strata. Census divisions are delineated by fine lines and numbered in fine script, waterfowl strata in thick lines and bold script.

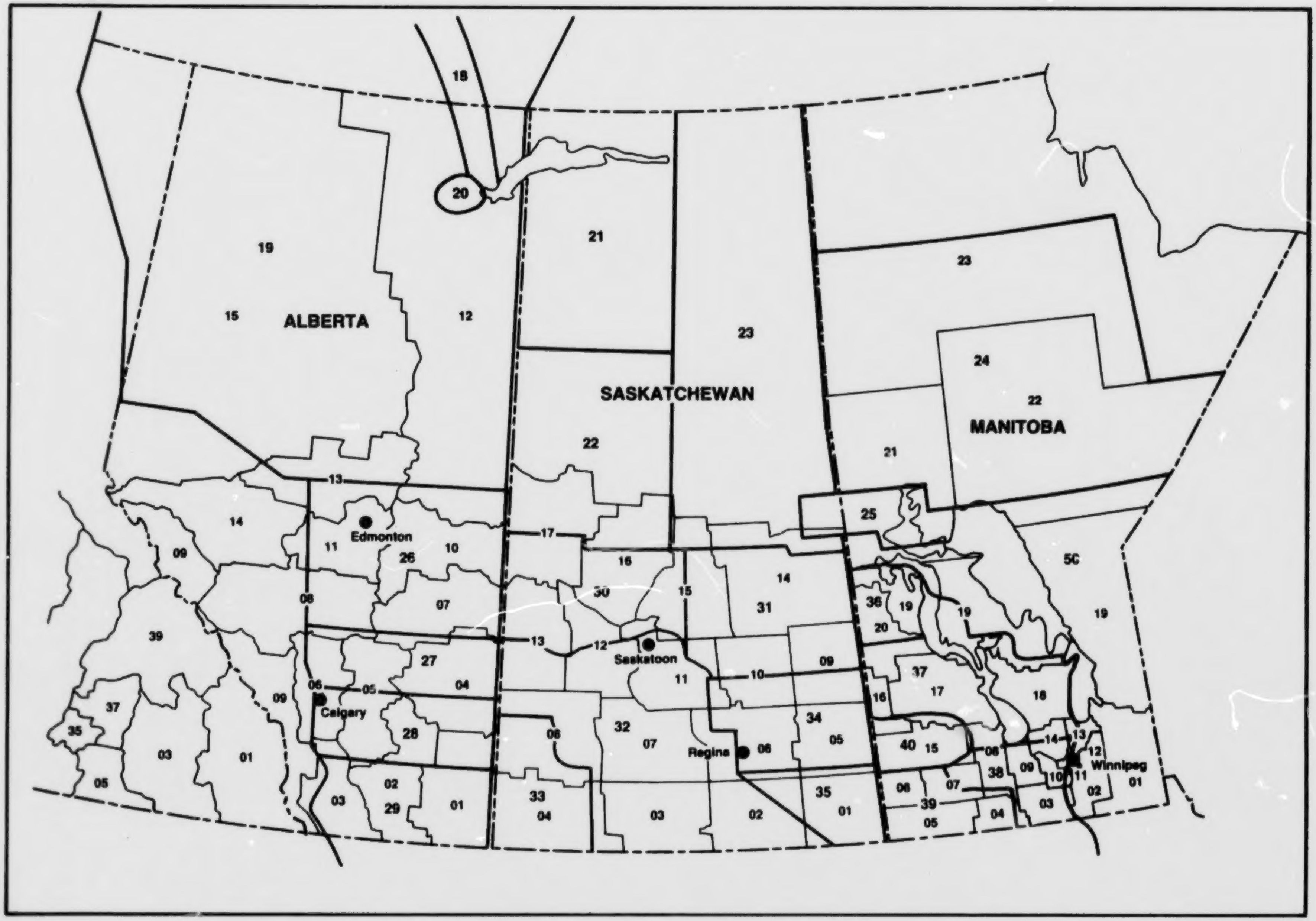

ALBERTA
SASKATCHEWAN
MANITOBA
Edmonton
Calgary
Saskatoon
Regina
Winnipeg
18
20
19
15
12
21
23
24
22
5C
25
13
17
14
11
26
10
09
08
07
16
30
15
31
36
19
20
39
27
04
12
06
05
28
37
35
03
01
02
29
33
32
11
10
09
34
40
38

Table 3.19. Estimated number of ducks (thousands) of all species breeding on land currently under crop[a] in the grasslands-parklands region of the prairie provinces.

Census Division	Numbers of Ducks (000's)		
	Alberta	Saskatchewan	Manitoba
1	159	475	N.A.[b]
2	140	338	N.A.
3	90	280	31
4	295	250	56
5	250	480	173
6	128	509	113
7	622	348	44
8	189	301	30
9	N.A.	358	21
10	671	425	16
11	228	388	3
12	N.A.	293	N.A.
13	N.A.	359	29
14	N.A.	511	27
15	N.A.	464	234
16	N.A.	406	51
17	N.A.	423	52
18	N.A.	N.A.	71
19	N.A.	N.A.	N.A.
20	N.A.	N.A.	36
Total	2,773	6,607	989

[a] This includes area in summer fallow

[b] N.A. - not applicable

Classes 1-3 (the assumed crop area) and, on the other hand, the area actually under crop or fallowed. The reader should recognize the high degree of data manipulation involved in obtaining these regional overlap estimates and the crop-specific data which follow. However, this was the only way available to us without a major reanalysis of the USFWS/CWS annual survey data. For a more accurate analysis, individual transects would have to be ascribed to a given census division or, better yet, to a set of crop conditions.

In fact, it should be possible, by slight modification of the yearly USFWS/CWS surveys, to investigate the direct impact of agricultural practices (including pesticide use) on breeding duck distribution and production (by using the brood survey data). To our knowledge, no one has attempted to correlate transect data with agricultural disturbance in any form. Given the generally recognized role of agriculture in reducing waterfowl production, we see this apparent lack of interest as a serious omission.

Assigning breeding ducks to a given crop type within any one census division is very difficult. In the absence of the data just mentioned, we must make yet further assumptions. Within the agricultural landscape, nesting preferentially takes place within the vegetative margin left intact around sloughs or in suitable "upland sites" such as idle grassland, field edges and roadsides. A limited amount of nesting by dabblers takes place in the cropland itself depending on crop phenology and the timing of cultivation practices (Figure 2.9). Based on literature values (Table 2.22), two crop types appear to differ from the others in their attractiveness to nesting ducks: tame hay has been reported to support nest densities 10-fold those found in other field crops whereas fallow is typically a third as attractive as cropped fields. It is useful here to remember that pastureland and its attending nesting duck populations have been excluded from this analysis. The nesting duck populations given in Table 3.19 are strictly those that nest in and around cropland and which are, therefore, more likely to be affected by agricultural spraying.

To calculate the number of ducks at risk with respect to any given spray operation, we must further distinguish between those birds actually nesting within the crop proper and those nesting in non-crop habitat. Diving ducks do not usually nest in upland habitat and must be treated separately from dabblers. We are forced to assume that their overlap with agriculture is similar to that of the dabblers but that they nest only in close association with the sloughs and that crop type is independant of slough density. Therefore, divers can be "assigned" to the various crop types on the basis of crop area only. The proportion of divers to total ducks is taken from the most representative strata as given in Appendix 3.1.

The "assignment" of dabblers to various crops is more complex. Whereas cropland and summer fallow in 1981 made up 65% of the total on-farm area (Table 3.10), a more conservative estimate of the non-crop area available to nesting ducks within an intensively cropped landscape is 10% (Higgins 1977, Sugden & Beyersbergen 1984). This area is comprised of those lands of poorer agricultural capability hidden within complexes of good agricultural land (Class 1-3) under crop or fallow (Section 3.3.1). Obviously, this 90:10 ratio will vary across the prairies, but in the absence of any hard data we will assume that it applies equally to all census divisions.

After dividing each census division into the seven major crop types and summer fallow, these areas are in turn divided into crop and "margin". For most census divisions, these 7 crops and summer fallow account for over 95% of the duck population which we estimate to breed in association with cropland. The only exceptions are a few districts in Manitoba (e.g. #3, 8 and 9) where other crop types (mainly row crops) account for 20-30% of the cropland and hence the estimated nesting duck population. Realistically, duck nesting in row crops is probably minimal. Nest density factors were used to apportion the nesting birds. The raw data on which these density factors were based were presented in Table 2.25. The actual values used in the computations are given in Table 3.20. All margin types were assumed to have the same attractiveness to nesting ducks.

The calculations used to estimate the number of dabblers and divers in the various agricultural crops are summarized in Figure 3.5. The results of these computations are detailed in Appendix 3.9 and summarized in Table 3.21.

Not suprisingly, the largest proportion of breeding ducks are associated with wheat since it is the dominant prairie crop and is broadly distributed throughout the region. Provided that the data of Duebbert and Kantrud (1974) on the attractiveness of forage crops are valid, tame hay achieves an importance, for dabbling ducks, that is totally out of proportion with its crop area. Tame hay and canola hold the largest numbers of nests of dabblers and divers, respectively, when expressed on a "per area" basis. The association of canola and diving ducks is explained by the co-correlation of the two with the parklands.

3.4.3 Inter-year variation in agriculture-waterfowl overlap and waterfowl production

So far, we have only considered average conditions for both waterfowl and agriculture. This belies the great between-year variations in both waterfowl and agricultural activity, mostly brought about by changes in the moisture regimes. We have also made the implicit assumption that all ducks are equal in their contribution to the waterfowl population. In fact, the reproductive output of ducks is known to vary widely between species as well as between years and habitats (Chapter 2). We have not so far made any

Table 3.20. Nest densities of ducks for various habitats. The value in brackets is the correction factor used for assigning duck nests to the available upland nesting habitat.

Habitat Type		Nest Densities (per ha)	Source
Grain crops and stubble	grain stubble	0.05	Duebbert & Kantrud 1974
	mulched stubble	0.02	Higgins 1977
	standing stubble	0.04	"
	growing grain	0.01	"
		mean = 0.03 (.03)	
Forage	hay	0.37	Duebbert & Kantrud 1974
	tame hay	0.25	"
		mean = 0.31 (.3)	
Fallow	summer fallow	0.01 (.01)	Higgins 1977
Margins (undisturbed land)	native grassland	0.11	Kirsh 1969
	undisturbed grass-legume cover	0.84	Duebbert & Kantrud 1974
	idle grassland	0.16	Higgins 1977
	idle pasture	0.15	"
	roadsides	0.36	"
	idle alfalfa	0.17	"
	dry wetland basins	0.14	"
		mean = 0.28 (.3)	

Table 3.21. Major crop areas in the grassland-parkland region of the prairie provinces and estimated numbers of ducks nesting within or in association with these crops (expressed as absolute numbers, densities and proportion of the totals nesting within these particular crop types)

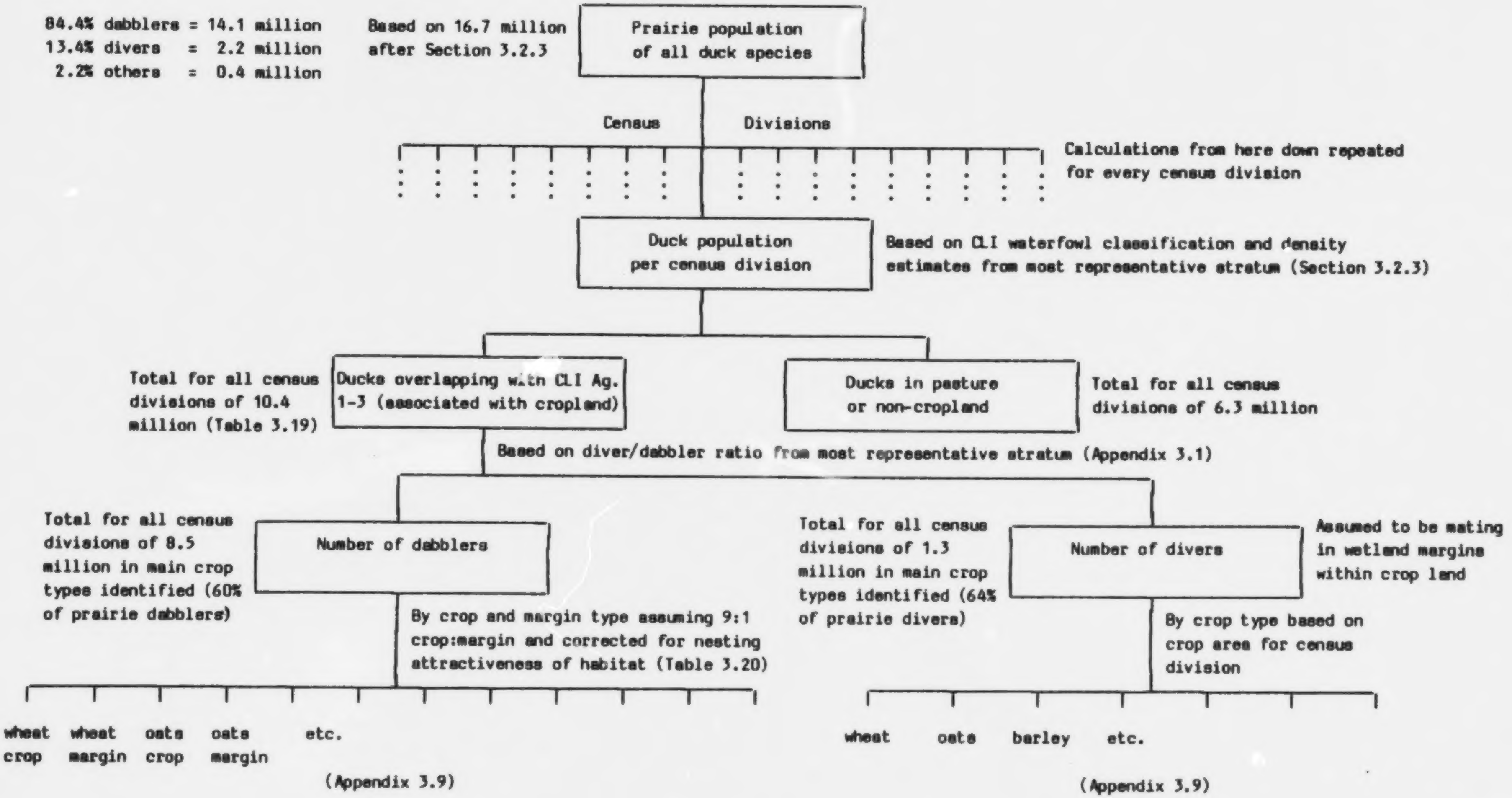
	Wheat	Oats	Barley	Rye	Flax	Canola	Tame Hay (alfalfa)	Summer Fallow	Total[b]
Area in grassland-parkland region (km^2)[a]	116,194	10,223	42,090	3,898	4,486	11,410	19,718	90,445	298,464
DABBLERS Number[c]	3,017,010	259,240	916,748	93,369	86,137	301,916	2,159,502	1,652,791	8,486,713
DABBLERS Proportion by crop type	35.5%	3.05%	10.8%	1.10%	1.01%	3.56%	25.4%	19.5%	100%
DABBLERS Density (per km^2 of crop)	26.0	25.4	21.8	24.0	19.2	26.5	110	18.3	28.4
DIVERS Number[c]	485,432	57,337	215,935	15,274	16,306	76,026	97,190	370,356	1,333,856
DIVERS Proportion by crop type	36.4%	4.30%	16.2%	1.15%	1.22%	5.70%	7.29%	27.8%	100%
DIVERS Density (per km^2 of crop)	4.18	5.61	5.13	3.92	3.63	6.66	4.93	4.09	4.47

[a] From 1981 Statistics Canada Census of Agriculture. Equals the sum of all census divisions falling within the grassland-parkland area as defined from waterfowl survey strata (Bellrose 1979)

[b] Excludes other minor crops (3.8% of total area in crop or summer fallow)

[c] From Appendix 3.9

Figure 3.5. Flow chart of calculations used to apportion ducks to specific crop types


84.4% dabblers = 14.1 million
13.4% divers = 2.2 million
2.2% others = 0.4 million
Based on 16.7 million
after Section 3.2.3
Prairie population
of all duck species
Census
Divisions
Calculations from here down repeated
for every census division
Duck population
per census division
Based on CLI waterfowl classification and density
estimates from most representative stratum (Section 3.2.3)
Total for all census
divisions of 10.4
million (Table 3.19)
Ducks overlapping with CLI Ag.
1-3 (associated with cropland)
Ducks in pasture
or non-cropland
Total for all census
divisions of 6.3 million
Based on diver/dabbler ratio from most representative stratum (Appendix 3.1)
Total for all census
divisions of 8.5
million in main crop
types identified (60%
of prairie dabblers)
Number of dabblers
By crop and margin type assuming 9:1
crop:margin and corrected for nesting
attractiveness of habitat (Table 3.20)
Total for all census
divisions of 1.3
million in main crop
types identified (64%
of prairie divers)
Number of divers
Assumed to be mating
in wetland margins
within crop land
By crop type based on
crop area for census
division
wheat
crop
wheat
margin
oats
crop
oats
margin
etc.
(Appendix 3.9)
wheat
oats
barley
etc.
(Appendix 3.9)

use of the annual brood counts conducted by the USFWS and the CWS (Section 3.1). There are many difficulties with these data and they are best used as rough indices of reproductive success only. Comparison between areas (a prerequisite for looking at the impact of a single factor such as pesticide impact on reproductive success) is difficult in view of the differential visibility of broods in different habitats. In this section we restrict our analysis to the brood counts obtained from the same transects over several years. We believe that this is a safer approach.

The impact of drought cycles on the agricultural and waterfowl resources is greater than the sum of the impacts on each individually. In periods of drought, the lowered water levels or the complete disappearance of water from large numbers of sloughs mean that many slough margins or entire sloughs are ploughed and placed under crop (Moulding 1983). The drought may be ephemeral, lasting only a summer or two, but the impact on the wetlands which had their basins disturbed may be much longer. To look at the between-year variation in breeding success, historical data from two strata (32 and 33) in southwestern Saskatchewan were isolated. These strata correspond to an area identified as being in chronic moisture stress as defined by the heat/precipitation ratio (Gage & Mukerji 1978). The numbers of ponds counted in May and July during the aerial surveys were used as an indication of water levels. The total number of breeding ducks in May and the total brood index were also noted. Data from the period 1955 to 1983 are presented in Table 3.22. For ease of visualization, each data set was transformed to percent deviations from its 29-year mean and was plotted against time (Figure 3.6). It is clear that the number of breeding ducks has been fluctuating, since the great drought of 1959-1962, in phase with the water levels as defined by the number of May ponds and, to a lesser extent, July ponds. For example, recent peaks in the number of breeding ducks in SW Saskatchewan have taken place in 1970-71, 1975, 1979 and 1982, all wet years following, in every case, a drought period of one or two breeding seasons. Contrary to what one would expect, the wet years are not associated with a high brood production defined as either the total number of broods or a productivity index. Leitch and Kaminski (1985) recently demonstrated a positive correlation between productivity and the number of July ponds for their pristine grassland study area in Saskatchewan. However, once years of drought were removed from their data set, the relationship between the number of adult breeders and breeding success was a negative one, at least for Mallards (R.M. Kaminski, pers. comm.). Density-dependant competition could be involved as could predator density. However, it is also important to consider the long term impact of drought on the ecology of sloughs. Draw-down is known to reduce the invertebrate population of sloughs in the short-term. Agricultural impact on the slough basin would exacerbate this impact (Krapu 1974b - see Chapter 2). Upon the return of wet conditions, a large number of sloughs may be present but they may not be able to support increased brood numbers. The habitat may only be suitable one or two years later when emergents have had a chance to reestablish (Chapter 2).

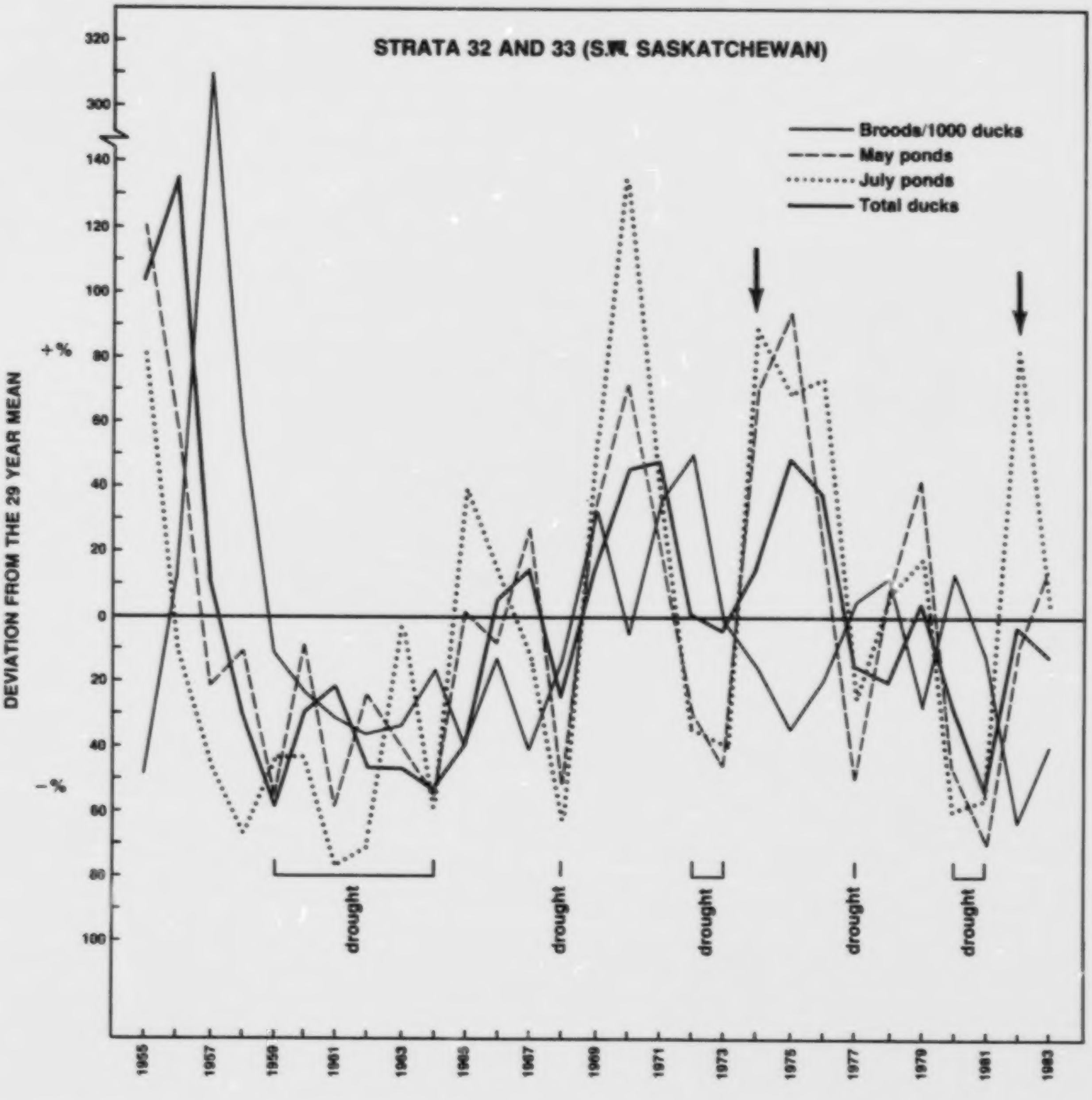
Since some insecticide uses are closely tied to the cycles of drought (e.g. grasshopper control programs), the ideas presented in this section will be important to the assessment of the overall spray impact on waterfowl.

Table 3.22. Total number of breeding ducks (thousands) and May and July ponds (thousands), total brood index and productivity index in Strata 32 and 33 combined for the period 1955-1983[a].

Year	Total Ducks (x 000)	May Ponds (x 000)	July Ponds (x 000)	Total Brood Index	Productivity Index (broods/1000 ducks)
1955	7218.8	1171.9	495.0	45,309	6.28
1956	8362.5	832.6	244.9	113,978	13.63
1957	3921.1	417.1	146.5	194,831	49.69
1958	2486.3	476.0	87.5	47,630	19.16
1959	1455.7	239.3	154.5	15,775	10.84
1960	2458.3	484.8	153.7	23,076	9.39
1961	2780.1	225.5	61.4	23,377	8.41
1962	1906.0	403.4	75.0	14,886	7.81
1963	1864.5	310.8	269.4	14,952	8.02
1964	1630.8	243.0	112.3	16,397	10.05
1965	2117.3	541.9	377.8	15,390	7.27
1966	3705.2	488.6	312.7	39,179	10.57
1967	4051.2	676.5	237.3	28,870	7.13
1968	2662.8	257.9	104.4	27,915	10.48
1969	4063.5	706.8	420.0	64,851	15.96
1970	5152.4	905.8	638.3	59,239	11.50
1971	5240.8	641.7	377.8	86,383	16.48
1972	3552.2	375.8	180.0	64,758	18.23
1973	3384.8	291.2	167.0	40,784	12.05
1974	4077.1	887.3	512.8	41,898	10.28
1975	5266.9	1022.2	460.2	41,540	7.89
1976	4865.3	665.3	469.5	47,709	9.81
1977	3001.1	268.5	202.9	38,289	12.76
1978	2813.1	573.5	290.6	38,236	13.59
1979	3660.7	751.5	321.1	32,147	8.78
1980	2594.8	287.4	112.3	35,436	13.66
1981	1631.0	159.1	119.3	17,672	10.84
1982	3451.6	477.8	494.6	15,390	4.46
1983	3124.1	609.2	281.6	22,912	7.33

[a] Source: U.S. Fish and Wildlife Service and Canadian Wildlife Service

Figure 3.6. May and July pond numbers, total ducks and a productivity index (broods/1,000 ducks) plotted as deviations from their 29-year mean (1955-1983) for strata 32 and 33 combined in SW Saskatchewan. Arrows indicate years of heavy insecticide spray - see section 4.2.4.2.


STRATA 32 AND 33 (S.W. SASKATCHEWAN)
Broods/1000 ducks
May ponds
July ponds
Total ducks
DEVIATION FROM THE 29 YEAR MEAN
+%
−%
320
300
140
120
100
80
60
40
20
0
20
40
60
80
100
drought
drought
drought
drought
drought
1955
1957
1959
1961
1963
1965
1967
1969
1971
1973
1975
1977
1979
1981
1983

Appendix 3.1. Adjusted breeding population means (for the 1970-1983 period) for all duck species surveyed annually by stratum (in thousands of ducks). Standard deviations are in parentheses. Total refers to strata 1-50.

	Stratum																							
Species	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	45	46	47	44	48	49	Total
Dabblers																									
Mallard	790.0 (238.2)	194.7 (95.2)	133.3 (49.0)	136.0 (43.8)	486.0 (158.6)	471.8 (113.0)	771.9 (224.8)	124.9 (46.8)	327.7 (80.3)	194.3 (70.9)	16.2 (5.5)	136.8 (34.9)	18.4 (8.1)	78.8 (26.7)	98.4 (17.4)	198.9 (63.7)	151.2 (44.5)	118.4 (32.1)	303.7 (127.7)	99.9 (32.3)	17.4 (14.8)	150.2 (54.5)	163.5 (65.0)	67.9 (31.3)	7797.8 (1229.4)
Black Duck	0.0	0.0	0.06 (0.24)	0.0	0.08 (0.3)	0.0	0.0	0.0	0.0	0.1 (0.2)	0.0	0.3 (0.7)	0.0	0.1 (0.3)	0.06 (0.2)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	85.0 (24.3)
Gadwall	194.1 (57.5)	55.9 (32.6)	44.7 (18.5)	32.7 (10.0)	93.6 (27.8)	71.2 (23.0)	282.0 (82.4)	29.2 (11.6)	52.4 (24.2)	48.5 (14.4)	2.0 (1.7)	16.5 (7.3)	1.0 (1.1)	24.05 (11.53)	19.4 (6.3)	61.6 (32.3)	20.2 (11.3)	32.4 (28.2)	177.9 (64.0)	81.0 (26.4)	4.25 (4.31)	33.9 (15.0)	91.9 (47.8)	23.2 (17.0)	1595.7 (169.4)
American Widgeon	175.0 (59.8)	65.7 (37.9)	52.6 (18.5)	55.9 (22.6)	150.2 (54.4)	93.9 (23.1)	315.5 (114.8)	46.2 (26.8)	58.5 (28.1)	36.3 (17.6)	2.6 (2.2)	16.0 (7.3)	1.2 (1.2)	10.1 (4.1)	13.0 (5.6)	90.9 (53.3)	38.1 (29.2)	24.8 (16.5)	111.3 (70.2)	44.0 (35.4)	4.6 (8.7)	51.95 (59.0)	52.2 (61.0)	27.0 (32.3)	3300.5 (410.9)
Green-winged Teal	194.4 (48.6)	34.8 (19.5)	22.4 (10.1)	17.1 (8.5)	41.5 (16.7)	46.4 (14.7)	71.2 (38.5)	9.0 (11.2)	75.1 (22.3)	63.8 (29.1)	5.1 (5.4)	37.8 (21.6)	4.5 (3.7)	28.5 (12.4)	26.6 (12.3)	36.7 (25.2)	29.6 (28.5)	26.8 (16.7)	27.4 (22.4)	11.5 (9.9)	1.7 (4.2)	24.7 (15.8)	21.7 (15.5)	10.2 (9.3)	2407.9 (354.0)
Blue-winged Teal	489.6 (197)	115.4 (66.5)	76.4 (31.2)	58.4 (27.0)	248.2 (107.3)	271.3 (125.0)	498.7 (208.3)	51.8 (43.0)	257.7 (118.2)	263.3 (108.0)	9.8 (6.2)	131.5 (85.8)	21.6 (12.9)	168.4 (108.6)	111.5 (61.1)	88.1 (56.7)	48.1 (30.1)	108.4 (67.9)	418.0 (221.0)	218.7 (96.1)	26.0 (23.6)	147.6 (92.0)	397.2 (198.2)	146.2 (103.9)	4960.4 (807.8)
Northern Shoveler	179.5 (62.8)	83.4 (51.1)	70.6 (26.2)	61.0 (36.8)	107.9 (37.3)	115.2 (45.5)	226.3 (97.1)	36.7 (20.8)	68.7 (25.2)	65.8 (32.9)	2.0 (1.2)	35.2 (17.8)	3.6 (2.6)	27.5 (14.4)	22.4 (8.4)	86.0 (52.1)	22.6 (12.1)	39.4 (18.9)	163.0 (88.7)	75.1 (61.2)	4.9 (6.3)	39.7 (20.3)	110.8 (92.6)	38.4 (37.1)	2137.3 (305.4)
Pintail	380.6 (185.3)	248.8 (105.0)	269.6 (132.7)	240.5 (131.5)	200.3 (95.5)	181.3 (83.9)	696.4 (366.6)	145.3 (76.2)	154.9 (71.0)	163.6 (103.6)	3.0 (3.3)	48.3 (38.1)	13.7 (11.0)	46.1 (26.9)	34.4 (19.0)	192.2 (101.0)	45.7 (19.9)	74.9 (50.1)	259.9 (165.0)	96.1 (52.1)	10.5 (14.2)	77.9 (42.8)	144.2 (117.1)	31.6 (21.3)	5720.8 (1173.0)
Subtotal	2411.2 (648.8)	798.8 (432.9)	669.7 (228.0)	599.7 (208.9)	1325.9 (400.0)	1251.2 (380.4)	2902.1 (903.4)	443.1 (186.3)	995.0 (285.6)	808.5 (306.7)	40.6 (14.8)	422.4 (150.6)	63.9 (29.7)	384.5 (166.1)	325.7 (87.8)	754.3 (255.0)	375.6 (108.3)	421.1 (131.7)	1461.3 (538.7)	626.4 (227.2)	69.2 (57.6)	526.0 (231.6)	981.5 (498.0)	344.4 (208.2)	27982.9 (3097.0)
Divers																									
Redhead	116.7 (68.9)	10.0 (7.0)	17.0 (19.6)	10.2 (5.9)	36.5 (32.4)	55.7 (16.0)	55.5 (22.7)	5.1 (5.2)	27.5 (11.6)	20.1 (14.1)	2.6 (3.1)	38.0 (18.6)	0.4 (0.7)	18.5 (10.3)	21.1 (9.2)	14.3 (13.1)	4.0 (8.4)	6.8 (9.3)	105.2 (50.3)	58.7 (28.3)	0.7 (1.8)	3.4 (3.7)	40.6 (33.0)	9.6 (6.9)	793.6 (164.8)
Canvasback	61.8 (23.8)	7.3 (4.6)	4.2 (2.8)	3.8 (2.9)	60.2 (19.4)	47.5 (14.6)	53.8 (19.5)	3.8 (2.1)	33.5 (18.7)	13.6 (8.9)	2.4 (2.7)	22.6 (13.1)	0.6 (1.1)	10.4 (5.7)	21.1 (9.7)	6.0 (4.2)	1.7 (2.1)	2.2 (3.4)	21.5 (11.1)	7.0 (3.5)	0.08 (0.16)	1.4 (2.0)	5.8 (4.5)	1.3 (1.2)	581.3 (106.5)
Scaup	347.2 (90.6)	44.9 '22.5)	51.4 (15.3)	36.1 (11.9)	139.8 (85.1)	85.9 (46.6)	155.8 (71.5)	22.6 (12.8)	43.8 (29.4)	35.9 (25.6)	9.4 (4.8)	47.3 (21.0)	0.9 (1.23)	37.3 (25.2)	37.1 (13.6)	45.6 (22.9)	16.4 (9.6)	20.3 (41.9)	44.7 (38.2)	17.4 (19.2)	0.7 (1.8)	13.3 (10.5)	16.8 (18.5)	9.6 (14.1)	7575.2 (970.8)
Ringneck	4.9 (2.9)	0.8 (1.1)	1.0 (1.2)	0.7 (0.7)	7.2 (4.1)	9.1 (6.0)	10.5 (10.5)	1.2 (1.7)	5.35 (5.0)	7.2 (5.5)	4.0 (5.5)	9.0 (7.2)	0.2 (0.35)	7.1 (6.2)	6.0 (5.7)	1.3 (3.8)	0.8 (1.3)	0.7 (1.7)	7.5 (8.8)	5.0 (5.7)	4.4 (14.3)	0.7 (1.8)	3.3 (4.4)	4.0 (6.7)	607.6 (156.9)
Goldeneye	14.5 (9.7)	1.0 (2.0)	0.7 (1.3)	1.0 (1.3)	14.0 (14.0)	8.4 (5.2)	0.9 (1.1)	0.2 (0.5)	0.3 (0.7)	0.5 (1.0)	8.8 (6.5)	12.6 (9.5)	0.0	0.04 (0.16)	1.7 (2.2)	0.6 (1.4)	0.5 (1.3)	0.09 (0.3)	0.2 (0.7)	0.0	0.0	0.2 (0.5)	0.1 (0.4)	0.1 (0.4)	758.6 (125.8)
Bufflehead	40.0 (11.5)	1.6 (2.3)	0.6 (0.8)	1.0 (1.3)	21.6 (8.6)	10.9 (4.2)	2.3 2.4	0.4 (0.6)	1.8 (1.2)	4.0 (1.9)	3.7 (1.6)	8.8 (5.1)	0.09 (0.2)	1.4 (1.2)	5.0 (2.5)	0.4 0.5	0.9 (1.1)	0.4 (0.7)	0.3 (0.6)	0.2 (0.4)	0.06 (0.2)	0.1 (0.4)	0.4 (0.5)	0.5 (1.1)	882.5 (146.7)
Subtotal	585.1 (150.5)	64.6 (28.4)	74.9 (29.0)	50.7 (16.3)	297.3 (121.0)	217.7 (63.2)	278.9 (105.2)	33.4 (14.9)	112.4 (43.3)	81.3 (45.5)	31.0 (16.7)	138.4 (42.1)	2.1 (1.8)	74.8 (30.5)	92.0 (20.5)	68.3 (32.1)	24.2 (15.8)	30.6 (47.0)	179.4 (78.5)	88.4 (46.7)	5.9 (17.7)	18.9 (11.5)	66.2 (49.6)	25.0 (24.3)	11198.4 (1065.1)
Miscellaneous																									
Oldsquaw	0.0	0.1 (0.6)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1430.8 (468.6)
Eider	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	22.6 (6.0)
Scoter	18.5 (10.3)	0.22 (0.35)	0.37 (0.71)	0.28 (0.57)	3.6 (3.1)	0.67 (0.88)	0.0	0.0	0.03 (0.1)	0.0	0.71 (0.8)	0.1 (0.2)	0.0	0.0	0.0	0.0	0.02 (0.08)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1468.9 (263.3)
Ruddy Duck	32.1 (20.9)	7.12 (7.38)	9.38 (7.89)	11.28 (20.4)	17.52 (9.5)	21.73 (16.74)	20.27 (14.33)	0.61 (1.76)	35.8 (15.3)	22.1 (37.3)	1.2 (2.8)	46.4 (27.9)	0.1 (0.4)	24.0 (26.3)	42.8 (25.4)	11.3 (12.3)	11.5 (13.3)	5.5 (15.6)	101.1 (65.5)	59.5 (64.6)	0.0	4.6 (5.9)	50.5 (54.4)	11.76 (16.5)	641.8 (281.0)
Mergansers	3.36 (3.17)	0.2 (0.37)	0.62 (0.69)	1.48 (1.49)	1.2 (1.3)	1.11 (1.31)	0.9 (1.35)	0.21 (0.45)	0.0	0.09 (1.0)	6.8 (8.2)	4.9 (6.9)	0.0	0.03 (0.07)	0.2 (0.3)	0.5 (0.8)	0.7 (0.8)	0.1 (0.3)	0.1 (0.4)	0.07 (0.2)	0.1 (10.4)	0.1 (0.3)	0.4 (1.0)	0.2 (0.4)	639.1 (181.9)
Subtotal	73.8 (17.3)	7.7 (7.6)	10.4 (8.6)	13.1 (22.3)	22.3 (11.8)	23.5 (17.1)	21.3 (15.3)	0.8 (1.8)	35.8 (15.9)	22.2 (38.8)	8.8 (8.0)	51.4 (31.3)	0.1 (0.4)	24.1 (27.4)	43.0 (26.3)	11.8 (13.1)	12.2 (14.1)	5.7 (16.2)	101.2 (68.0)	59.5 (67.1)	0.1 (0.4)	4.7 (6.2)	51.0 (56.4)	11.9 (17.3)	4203.8 (813.4)
Total	3070.2 (613.3)	871.2 (447.6)	755.0 (229.9)	663.5 (224.9)	1645.5 (388.2)	1492.5 (316.1)	3202.3 (929.7)	477.4 (192.0)	1163.3 (311.3)	912.0 (330.9)	80.4 (30.3)	612.2 (166.2)	66.1 (30.0)	483.4 (168.7)	460.7 (79.5)	834.3 (269.3)	412.0 (110.2)	457.4 (179.3)	1761.9 (582.2)	774.3 (303.5)	75.2 (67.7)	549.6 (237.8)	1098.7 (535.3)	381.4 (227.2)	43385.2 (3366.1)
American Coot	238.61 (97.6)	26.81 (15.61)	33.96 (30.35)	56.24 (62.0)	130.3 (76.27)	189.63 (113.66)	237.56 (153.16)	15.8 (12.5)	138.8 (91.4)	96.1 (50.4)	10.5 (10.5)	85.2 (50.2)	9.0 (7.8)	102.9 (60.4)	74.1 (32.5)	50.6 (56.8)	31.5 (50.1)	48.7 (52.4)	237.2 (155.5)	133.6 (120.5)	4.7 (5.9)	18.1 (18.7)	99.9 (61.1)	46.8 (48.3)	2449.3 (625.1)

Appendix 3.2. Canadian and U.S. prairie (grassland and parkland) subtotals (in thousands of ducks) of mean estimated breeding duck populations (1970-1983)[a].

Species	Canadian Prairie Estimate	% of Total U.S.F.W. Surv. Pop.	Northern U.S. Prairie Estimate	% of Total U.S.F.W. Surv. Pop.	N.American Prairie Estimate	% of Total U.S.F.W. Surv. Pop.
Dabblers						
Mallard	3983.2	51.1	1271.1	16.3	5254.3	67.4
Black Duck	0.68	0.8	0.0	0.0	0.7	0.8
Gadwall	968.0	60.7	526.4	33.0	1494.4	93.8
American Widgeon	1092.7	33.1	464.8	14.1	1557.5	47.2
Green-winged Teal	678.2	28.2	190.3	7.9	868.5	36.1
Blue-winged Teal	2746.6	55.6	1594.3	32.3	4340.9	87.9
Northern Shoveler	1145.8	53.6	579.9	27.1	1725.7	80.7
Pintail	2826.6	49.4	933.0	16.3	3759.6	65.7
Subtotal	13442.3	48.0	5559.8	19.9	19002.1	67.9
Divers						
Redhead	452.9	57.1	243.3	30.7	696.2	87.7
Canvasback	346.6	59.6	46.1	7.9	392.7	67.5
Scaup	1093.4	14.4	184.8	2.4	1278.2	16.9
Ringneck	74.2	12.2	27.7	4.5	101.9	16.9
Goldeneye	64.6	8.5	1.8	0.2	66.4	8.7
Bufflehead	103.2	11.7	3.8	0.4	107.0	12.1
Subtotal	2134.6	19.1	506.9	4.5	2641.5	23.6
Miscellaneous						
Oldsquaw	0.0	0.0	0.0	0.0	0.0	0.0
Eider	0.0	0.0	0.0	0.0	0.0	0.0
Scoter	24.5	1.7	0.02	0.0	24.5	1.7
Ruddy Duck	312.4	48.7	255.8	39.8	568.2	88.5
Mergansers	21.0	3.3	2.3	0.4	23.3	3.6
Subtotal	358.3	8.5	258.1	6.1	616.4	14.7
American Coot	1445.3	59.0	671.1	27.4	2116.4	86.4

[a] Source: Appendix 3.1

Appendix 3.3. Provincial and state subtotals (in thousands of ducks) of mean estimated breeding duck populations (1970-1983)[a].

Species	S. Alta. Str. 26, 27, 28, 29	% of Total U.S.F.W. Surv. Pop.	S. Sask. Str. 30, 31, 32, 33, 34, 35	% of Total U.S.F.W. Surv. Pop.	S.W. Man. Str. 36, 37, 38, 39, 40	% of Total U.S.F.W. Surv. Pop.
Dabblers						
Mallard	1260	16.2	2374.6	30.4	348.6	4.5
Black Duck	0.06	0.07	0.18	0.2	0.44	0.5
Gadwall	327.4	20.5	576.9	36.2	63.75	4.0
American Widgeon	349.2	10.6	700.6	21.2	42.9	1.3
Green-winged Teal	268.7	11.2	307.0	12.7	102.5	4.3
Blue-winged Teal	739.8	15.0	1564.0	31.7	442.8	9.0
Northern Shoveler	394.5	18.5	660.6	30.9	90.7	4.2
Pintail	1139.5	19.9	1541.6	26.9	145.5	25.4
Subtotal	4479.4	16.0	7725.8	27.6	1237.1	4.4
Divers						
Redhead	153.9	19.4	218.4	27.5	80.6	10.2
Canvasback	77.1	13.3	212.4	36.5	57.1	9.8
Scaup	477.6	6.3	483.8	6.4	132.0	1.7
Ringneck	7.4	1.2	40.5	6.7	26.3	4.3
Goldeneye	17.2	2.3	24.3	3.2	23.1	3.0
Bufflehead	43.2	4.9	41.0	4.6	19.0	2.1
Subtotal	775.3	6.9	1021	9.1	338.3	3.0
Miscellaneous						
Oldsquaw	0.0	0.0	0.0	0.0	0.0	0.0
Eider	0.0	0.0	0.0	0.0	0.0	0.0
Scoter	19.4	1.3	4.3	0.3	0.8	0.05
Ruddy Duck	79.9	12.4	118.0	18.4	114.5	17.8
Mergansers	5.6	0.9	3.5	0.5	11.9	1.9
Subtotal	105	2.5	125.9	3.0	127.4	3.0
Total	5359.9	12.3	8873.0	20.4	1702.8	3.9
American Coot	355.6	14.5	808.0	33.0	281.7	11.5

Appendix 3.3. (cont'd)

Species	N.E. Mont. Str. 41, 42	% of Total U.S.F.W. Surv. Pop.	N. Dak. Str. 43, 45 46, 47	% of Total U.S.F.W. Surv. Pop.	S. Dak. Str. 44, 48, 49	% of Total U.S.F.W. Surv. Pop.
Dabblers						
Mallard	350.1	4.5	539.4	6.9	381.6	4.9
Black Duck	0.0	0.0	0.0	0.0	0.0	0.0
Gadwall	81.8	5.1	295.6	18.5	149.0	9.3
American Widgeon	149.0	4.5	184.7	5.6	131.1	4.0
Green-winged Teal	66.3	2.7	67.4	2.8	56.6	2.3
Blue-winged Teal	136.2	2.8	767.1	15.5	691.0	14.0
Northern Shoveler	108.6	5.1	282.4	13.2	188.9	8.8
Pintail	237.9	4.2	441.4	7.7	253.7	4.4
Subtotal	1129.9	4.0	2578	9.2	1851.9	6.6
Divers						
Redhead	18.3	2.3	171.4	21.6	53.6	6.7
Canvasback	7.7	1.3	30.7	5.3	7.7	1.3
Scaup	62.0	0.8	83.1	1.1	39.7	0.5
Ringneck	2.1	0.3	17.6	2.9	8.0	1.3
Goldeneye	1.1	0.1	0.3	0.04	0.4	0.05
Bufflehead	1.3	0.1	1.5	0.2	1.0	0.1
Subtotal	92.5	0.8	304.3	2.7	110.1	1.0
Miscellaneous						
Oldsquaw	0.0	0.0	0.0	0.0	0.0	0.0
Eider	0.0	0.0	0.0	0.0	0.0	0.0
Scoter	0.02	0.0	0.0	0.0	0.0	0.0
Ruddy Duck	22.8	3.5	166.1	25.9	66.9	10.4
Mergansers	1.2	0.2	0.4	0.06	0.7	0.1
Subtotal	24.0	0.6	166.5	4.0	67.6	1.6
Total	1246.3	2.9	3048.8	7.0	2029.7	4.7
American Coot	82.1	3.3	424.2	17.3	164.8	6.7

[a] Source: Appendix 3.1

Appendix 3.4. Mean estimated breeding duck population (in thousands of ducks) (1970-1983) broken down by major vegetation zone

Species	Parkland Total (Str. 26, 30, 31, 34, 38, 40)	% of Total U.S.F.W. Surv. Pop.	Can. Mixed Prairie (Str. 27, 28, 32, 35, 39)	% of Total U.S.F.W. Surv. Pop.	U.S. Mixed Prairie (Str. 45, 46 48	% of Total U.S.F.W. Surv. Pop.
Dabblers						
Mallard	2198.3	28.2	1373.0	17.6	567.1	7.3
Black Duck	0.1	0.1	0.3	0.3	0.0	0.0
Gadwall	431.7	27.1	455.9	28.6	350.8	22.0
American Widgeon	491.8	14.9	480.2	14.5	207.5	6.3
Green-winged Teal	388.5	16.1	220.7	9.2	60.6	2.5
Blue-winged Teal	1339.9	28.3	1095.2	22.2	1033.9	20.9
Northern Shoveler	497.3	23.3	513.6	24.0	348.9	16.3
Pintail	965.2	16.9	1424.3	24.9	500.2	8.7
Subtotal	6372.9	22.8	5563.6	19.9	3069.2	11.1
Divers						
Redhead	275.9	34.8	121.1	15.3	204.5	25.8
Canvasback	224.7	38.6	89.3	15.4	33.5	5.8
Scaup	654.7	8.6	325.3	4.3	78.9	1.0
Ringneck	32.7	5.4	26.6	4.4	15.8	2.6
Goldeneye	38.9	5.1	3.1	0.4	0.3	0.04
Bufflehead	79.4	9.0	9.9	1.1	1.4	0.2
Subtotal	1306.6	11.7	574.5	5.1	334.0	3.0
Miscellaneous						
Oldsquaw	0.0	0.0	0.0	0.0	0.0	0.0
Eider	0.0	0.0	0.0	0.0	0.0	0.0
Scoter	22.8	1.5	0.6	0.04	0.0	0.0
Ruddy Duck	170.0	26.5	82.9	12.9	211.1	32.9
Mergansers	5.8	0.9	1.8	0.3	0.5	0.08
Subtotal	198.5	4.7	85.7	2.0	211.7	5.0
Total	7878.3	18.2	6223.9	14.3	3614.9	8.3
American Coot	780.2	31.8	497.3	20.3	470.7	19.2

Appendix 3.4. (cont'd)

Species	Mixed Prairie Total	% of Total U.S.F.W. Surv. Pop.	Can. Shortgrass (Str. 29, 33)	% of Total U.S.F.W. Surv. Pop.	U.S. Shortgrass (Str. 41, 42, 43, 44	% of Total U.S.F.W. Surv. Pop.
Dabblers						
Mallard	1940.1	24.9	258.9	3.3	618.7	7.9
Black Duck	0.3	0.3	0.0	0.0	0.0	0.0
Gadwall	806.7	50.6	61.9	3.9	148.1	9.3
American Widgeon	687.7	20.8	102.1	3.1	225.7	6.8
Green-winged Teal	281.3	11.7	26.1	1.8	117.8	4.9
Blue-winged Teal	2129.1	43.1	110.2	22.3	388.2	78.6
Northern Shoveler	862.5	40.3	97.7	4.6	187.7	8.8
Pintail	1924.5	33.6	385.8	6.7	390.7	6.8
Subtotal	8632.8	30.8	1042.8	3.7	2077.0	7.4
Divers						
Redhead	325.6	41.0	15.3	1.9	28.5	3.6
Canvasback	122.8	21.1	7.6	1.3	11.3	1.9
Scaup	404.2	5.3	56.7	0.7	95.6	1.3
Ringneck	42.4	7.0	1.9	0.3	3.5	0.6
Goldeneye	3.4	0.4	1.2	0.2	1.4	0.2
Bufflehead	11.3	1.3	1.4	0.2	1.8	0.2
Subtotal	908.5	8.1	84.1	0.7	142.0	1.3
Miscellaneous						
Oldsquaw	0.0	0.0	0.0	0.0	0.0	0.0
Eider	0.0	0.0	0.0	0.0	0.0	0.0
Scoter	0.6	0.04	0.3	0.02	0.02	0.0
Ruddy Duck	294.0	45.8	11.9	1.8	32.9	5.1
Mergansers	2.3	0.4	1.7	0.3	1.4	0.2
Subtotal	297.4	7.1	13.9	0.3	34.4	0.8
Total	9838.8	22.7	1140.9	2.6	2253.3	5.2
American Coot	968.0	39.5	72.0	2.9	148.9	6.1

Appendix 3.4. (cont'd)

Species	Shortgrass Total	% of Total U.S.F.W. Surv. Pop.	Tallgrass Prairie (Str. 47, 49)	% of Total U.S.F.W. Surv. Pop.
Dabblers				
Mallard	877.6	11.2	85.3	1.1
Black Duck	0.0	0.0	0.0	0.0
Gadwall	210.0	13.2	27.4	1.7
American Widgeon	327.8	9.9	31.6	9.6
Green-winged Teal	143.9	6.0	11.9	0.5
Blue-winged Teal	498.4	10.1	172.2	3.5
Northern Shoveler	285.4	13.3	43.3	2.0
Pintail	776.5	13.6	42.1	0.7
Subtotal	3119.8	11.1	413.6	1.5
Divers				
Redhead	43.8	5.5	10.3	1.3
Canvasback	18.9	3.2	1.34	0.2
Scaup	152.3	2.0	10.3	0.1
Ringneck	5.4	0.9	8.4	1.4
Goldeneye	2.6	0.3	0.1	0.01
Bufflehead	3.2	0.4	0.56	0.06
Subtotal	226.1	2.0	30.9	0.3
Miscellaneous				
Oldsquaw	0.0	0.0	0.0	0.0
Eider	0.0	0.0	0.0	0.0
Scoter	0.32	0.02	0.0	0.0
Ruddy Duck	44.8	7.0	11.8	1.8
Mergansers	3.1	0.5	0.3	0.05
Subtotal	48.3	1.1	12.0	0.3
Total	3394.2	7.8	456.6	1.0
American Coot	220.9	9.0	51.5	2.1

Source: Appendix 3.1 following landform classification of Bellrose (1979)

Appendix 3.5. Comparison of the full Canada Land Inventory (CLI) data base with the more restricted data base corresponding to the prairies (grasslands and parklands) following Patterson (1978)

This appendix compares the full CLI data base (Lands Directorate 1978b) with the geographically restricted data set compiled for Patterson (1978) by the Canada Lands Directorate. Table A compares the land areas included in the two data sets and shows the close correspondence between Patterson's restricted CLI area with the prairie waterfowl strata (identified as either parkland or grassland by Bellrose (1979) but also including strata 36 and 37 in Manitoba which are generally considered to be boreal). The scope of the "restricted" CLI data base can be more accurately defined as the "southern prairie provinces", but for the purpose of the quantitative analysis of waterfowl or agricultural potential, it will be equated with the grassland-parkland area as envisioned by Bellrose and others. Table A also presents the total area farmed and total provincial land area (Statistics Canada 1981). Of note is the extended coverage of the full CLI data base for Alberta. This is a result of the inclusion of the Peace River region, which is intensively developed notwithstanding its higher latitude, in this data base. The Peace River region has been excluded from our analysis since there is no adequate match between it and a waterfowl stratum. This omission will be discussed in the text. Table B shows that Patterson's restricted CLI area includes most of the best (Class 1-3) waterfowl habitat of the prairie provinces. The slightly greater discrepancy for Alberta is again accounted for by the Peace River district. Table C expresses the same data (from both the full and restricted CLI data set) as percentages of the total waterfowl habitat within designated capability classes.

Appendix 3.5 (cont'd)

Table A. Comparison of the full CLI data base, Patterson's (1978) restricted data base and prairie area covered by waterfowl surveys.

Province	Area Covered by Restricted CLI Data (Paterson 1978)		Total Area Covered by CLI Data		Total Prairie Area Covered by USFWS & CWS Surveys[a]		Total Area of Farms in 1981[b]		Total Land Area[b]
	km^2	%[c]	km^2	%[c]	km^2	%[c]	km^2	%[c]	km^2
Alberta	150,958	23.6	478,745	75.0	165,362	25.9	191,085	29.9	638,230
Saskatchewan	302,729	56.2	357,118	62.6	287,801	50.5	259,471	45.5	570,113
Manitoba	122,101	22.3	216,321	39.5	100,302	18.3	76,159	13.9	547,705
Prairies	575,788	32.9	1,052,184	59.9	553,465	31.5	526,720	30.0	1,756,048

[a] Those waterfowl strata identified as parkland or grassland by Bellrose (1979) as well as Strata 36 and 37 which are generally considered to be boreal

[b] Data from the 1981 census of Canada (Statistics Canada)

[c] Percent of total land area

Appendix 3.5 (cont'd)

Table B. Area (km^2), by waterfowl class, in Patterson's restricted data base and in the full CLI data base.

Waterfowl Class	Alberta		Saskatchewan		Manitoba		Total	
	Area Covered by Patterson's (1978) Restricted CLI Data	Total Area Covered By CLI Data	Area Covered by Patterson's (1978) Restricted CLI Data	Total Area Covered By CLI Data	Area Covered by Patterson's (1978) Restricted CLI Data	Total Area Covered By CLI Data	Area Covered by Patterson's (1978) Restricted CLI Data	Total Area Covered By CLI Data
1	16,559	17,617	6,405	6,471	1,074	1,113	24,039	25,202
2	5,803	7,529	25,870	25,997	4,316	4,638	35,989	38,165
3	17,314	20,707	49,737	50,032	10,246	10,224	77,297	80,963
4	18,900	28,456	78,410	79,235	20,882	22,239	118,192	129,930
5	32,136	56,556	85,947	93,685	37,158	43,830	155,242	194,071
1-2	22,362	25,146	32,275	32,468	5,390	5,751	60,028	63,367
1-3	39,677	45,853	82,012	82,501	15,636	15,975	137,325	144,330
4-5	51,036	85,012	164,357	172,920	58,041	66,069	273,434	324,000
1-5	90,713	130,865	246,370	255,420	73,677	82,044	410,760	468,330

Appendix 3.5 (cont'd)

Table C. Patterson's (1978) restricted area and the full area covered by the CLI expressed as percentages of the total waterfowl habitat within designated capability classes.

CLI Waterfowl Class	Alberta		Saskatchewan		Manitoba		Total	
	% in Patterson's Restricted Area	% in full CLI Area	% in Patterson's Restricted Area	% in full CLI Area	% in Patterson's Restricted Area	% in full CLI Area	% in Patterson's Restricted Area	% in full CLI Area
1-2	14.8	5.2	10.7	9.0	4.4	2.7	10.4	6.0
3	11.5	4.3	16.4	14.0	8.4	4.7	13.4	7.7
1-3	26.3	9.5	27.1	23.1	12.8	7.4	23.8	13.7
4-5	33.8	17.8	54.2	48.4	47.5	30.5	47.5	30.8
1-5	60.1	27.3	81.4	71.5	60.3	37.9	71.3	44.5

Appendix 3.6. Land areas (in km^2) classified by Agricultural Class within the parkland-grassland area as defined by Patterson (1978) compared to areas classified under the full CLI data base.

Agricultural Class	Alberta		Saskatchewan		Manitoba		Total	
	Parkland and Grasslands	CLI	Parkland and Grasslands	CLI	Parkland and Grasslands	CLI	Parkland and Grasslands	CLI
1	7,343	7,989	11,668	12,564	2,126	2,126	21,137	22,679
2	27,944	41,254	71,807	73,983	29,276	29,914	129,027	145,151
3	35,357	63,671	101,921	104,632	22,138	24,981	159,416	193,284
4	34,790	100,327	37,731	42,357	25,926	28,363	98,447	171,047
1-3	70,644	112,914	185,396	191,179	53,540	57,021	309,580	361,114
1-4	105,434	213,241	223,127	233,536	79,466	85,384	408,027	532,161
5-7+0	45,524	265,504	79,602	123,582	42,635	130,937	167,761	520,023
TOTAL	150,958	478,745	302,729	357,118	122,101	216,321	575,788	1,052,184

Appendix 3.7. "Present land use" compilation of data classified under the CLI (1965 data).
"Organic" refers to muck soils which are classed apart from other soils.

		Class 1			Class 2			Class 3			Class 4			Class 5			Class 6			Class 7			Organic			Total	
Prov.	Land Use[b]	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class
ALBERTA	Woodland (T&U)	7,122	1	1	319,539	8	2	1,460,704	24	8	4,601,006	53	25	5,299,779	62	29	1,440,361	45	8	1,275,562	81	7	3,659,141	86	20	18,063,214	49
	Pasture & Forage (P)	50,278	6	6	202,946	5	23	255,703	4	29	168,611	2	19	114,391	1	13	50,216	2	6	6,004	11	1	24,182	1	3	872,331	2
	Grazing & Range-land (K)	74,576	9	1	518,794	14	7	1,114,767	18	15	1,805,499	21	25	2,380,857	28	33	1,160,122	36	16	125,295	8	2	94,604	2	1	7,282,514	20
	Cropland (A)	646,048	82	6	2,756,859	72	28	3,230,660	53	33	2,074,940	24	20	590,265	7	6	385,835	16	4	55,194	3	1	146,599	3	1	9,886,400	27
	Other (M,B,O, L,S,H,G, E)	8,492	1	1	40,632	1	4	47,034	1	5	90,362	1	10	142,797	2	15	183,161	6	20	102,856	7	11	311,934	7	34	927,268	2
	Total	786,516	40	2	3,838,770	31	10	6,108,868	34	16	8,740,418	59	24	8,536,089	44	23	3,219,695	39	9	1,564,911	60	4	4,236,460	41	11	37,031,727	42
SASKATCHEWAN	Woodland (T&U)	53,984	5	1	764,408	13	11	1,362,742	14	19	752,234	20	10	1,741,183	20	24	1,081,687	37	15	2,873	20	1	1,456,227	85	20	7,215,338	22
	Pasture & Forage (P)	2,615	1	1	25,388	1	12	44,569	1	21	35,033	1	16	88,414	1	41	19,598	1	9	150	1	1	469	1	1	216,236	1
	Grazing & Range-land (K)	63,978	6	1	600,886	10	8	1,269,931	13	17	921,188	24	13	3,085,463	36	42	1,332,467	46	18	6,629	47	1	34,996	2	1	7,315,538	22
	Cropland (A)	873,311	87	5	4,448,761	75	25	6,662,720	71	37	2,007,445	53	11	3,436,815	40	19	378,068	13	2	2,596	18	1	9,513	1	1	17,819,229	54
	Other (M,B,O, L,S,H,G, E)	5,790	1	1	33,535	1	6	77,572	1	13	54,764	1	9	147,487	2	24	68,909	2	11	1,921	14	1	211,036	12	35	601,014	2
	Total	999,678	51	3	5,872,978	48	17	9,417,534	52	23	3,770,664	25	11	8,499,362	44	26	2,880,729	35	9	14,169	1	1	1,712,241	16	5	33,167,355	38

Appendix 3.7 (cont'd)

Prov.	Land Use[b]	Class 1			Class 2			Class 3			Class 4			Class 5			Class 6			Class 7			Organic			Total	
		Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class	% of all classes	Area (ha)	% of class
MANITOBA	Woodland (T&U)	6,885	4	1	206,634	8	2	553,187	23	6	1,331,561	56	14	1,284,186	57	14	1,360,917	75	17	737,394	71	8	3,319,599	79	38	9,200,363	53
	Pasture & Forage (P)	3,583	2	1	90,806	4	16	167,160	7	30	157,353	7	28	106,220	5	19	20,663	1	4	5,099	1	1	14,598	1	3	565,482	3
	Grazing & Range-land (K)	1,032	1	1	82,032	3	9	125,908	5	14	225,823	9	25	295,991	13	32	75,553	4	8	23,220	2	2	88,565	2	10	918,132	5
	Cropland (A)	148,874	92	3	2,111,300	83	41	1,349,924	63	30	569,111	24	11	403,449	18	8	256,658	12	5	37,697	4	1	41,[illegible]43	1	1	5,118,156	30
	Other (M,B,O, L,S,H,G, E)	2,125	1	1	39,252	2	3	44,122	2	3	106,106	4	7	146,836	7	10	171,214	8	11	230,605	22	15	782,405	18	51	1,522,665	9
	Total	162,499	8	1	2,530,024	21	15	2,440,301	14	14	2,389,954	16	14	2,236,682	12	13	2,085,005	25	12	1,034,023	40	6	4,446,310	43	26	17,324,798	20
TOTAL	Woodland (T&U)	67,991	3	1	1,290,581	10	4	3,376,633	19	10	6,684,801	45	19	8,325,148	43	24	4,082,965	50	12	2,015,829	77	6	8,634,967	83	25	34,478,915	39
	Pasture & Forage (P)	56,476	3	3	319,140	3	19	467,432	3	28	360,997	2	22	309,025	2	19	90,477	1	5	11,253	1	1	39,249	1	2	1,654,049	2
	Grazing & Range-land (K)	139,586	7	1	1,201,712	10	8	2,510,606	14	16	2,952,510	20	19	5,770,311	30	37	2,568,142	31	17	155,152	6	1	218,165	2	1	15,516,184	18
	Cropland (A)	1,668,233	86	5	9,316,920	76	28	11,443,308	64	35	4,651,496	31	14	4,430,529	23	13	1,020,561	12	3	95,487	4	1	197,255	2	1	32,823,785	37
	Other (M,B,O, L,S,H,G, E)	16,407	1	1	113,419	1	6	168,728	1	5	251,232	2	8	437,120	2	14	423,284	5	14	335,382	13	11	1,305,375	13	43	3,050,967	3
	Total	1,948,693		2	12,241,772		14	17,966,703		20	14,901,036		17	19,272,133		22	8,185,429		9	2,613,103		3	10,395,011		12	87,523,880	

[a] From Lands Directorate 1986
[b] Symbols for Classification for Present Land Use:
(See attached)

URBAN

Land use for urban and associated non-agricultural purposes.

	Symbol
1. Built-up Area (Parks and other open spaces within built-up areas are included.)	B
2. Mines, Quarries, Sand and Gravel Pits (Land used for the removal of earth materials.)	E
3. Outdoor Recreation (Golf courses, parks, beaches, summer cottage areas, game preserves and historical sites.)	O

AGRICULTURAL LANDS

1. Horticulture, Poultry and Fur Operations Land used for intensive cultivation of vegetables and small fruits including market gardens, nurseries, flower and bulb farms, and sod farms. Large scale commercial fur and poultry farms are also included because of their specialized agricultural nature.	H
2. Orchards and Vineyards Land used for the production of tree fruits, hops and grapes.	G
3. Cropland Land used for annual field crops: grain oilseeds, sugar beets, tobacco, potatoes, field vegetables, associated fallow, and land being cleared for field crops.	A
4. Improved Pasture and Forage Crops Land used for improved pasture or for the production of hay and other cultivated fodder crops including land being cleared for these purposes.	P
5. Rough Grazing and Rangeland	
(a) Areas of natural grasslands, sedges, herbaceous plants and abandoned farmland whether used for grazing or not. Bushes and trees may cover up to 25 per cent of the area. If in use, intermittently-wet, hay lands (sloughs or meadows) are included.	K
(b) Woodland grazing: If the area is actively grazed and no other use dominates, in some grassy, open woodlands, bushes and trees may somewhat exceed 25 per cent cover.	

WOODLAND	**Symbol**
Land covered with tree, scrub or bush growth, including:	T
1. **Productive Woodland** Wooded land with trees having over 25 per cent canopy cover and over approximately 20 feet in height. Plantations and artificially reforested areas are included regardless of age.	
2. **Non-Productive Woodland** Land with trees or bushes exceeding 25 per cent Crown cover, and shorter than approximately 20 feet in height. Much cut-over and burned-over land is included.	U
WETLAND	
Swamp, Marsh or Bog Open wetlands except those which frequently dry up, and show evidence of grazing or hay cutting. (See K Agricultural Lands.)	M
UNPRODUCTIVE LAND	
Land which does not, and will not, support vegetation. e.g. eroded soil or rock and active depositional features.	
1. **Sand** (Sand bars, sand flats, dunes, beaches.)	S
2. **Rock and Other Unvegetated Surfaces** (Rock barrens, badlands, alkali flats, gravel bars, eroded river banks, mine dumps.)	L

Appendix 3.8.

Agriculture-Waterfowl Land Capability Cross Tabulation

Southeastern Alberta

By primary CLI agriculture and waterfowl classification (in km^2)

	CLI Agriculture									
Waterfowl Class	1	2	3	4	5	6	7	8	0[a]	Total
1	391.66	3,858.45	5,468.31	4,373.03	2,003.90	434.13	8.73	6.22	15.00	16,559.43
2	52.84	720.53	2,250.59	1,568.03	894.68	296.81	8.37	1.29	9.92	5,803.06
3	795.51	3,868.76	5,125.89	4,035.17	2,542.74	801.47	71.43	67.70	5.59	17,314.26
4	1,479.35	3,939.70	4,936.56	4.557.70	2,807.00	1,004.40	105.52	55.30	14.45	18,898.98
5	2,443.99	4,933.86	6,213.01	8,044.51	7,359.77	1,837.62	46.57	1,144.98	111.76	32,136.07
6	2,149.81	10,397.85	10,998.83	11,726.09	11,581.88	6,003.86	491.66	1,198.93	75.78	54,624.69
7	25.41	216.52	358.92	484.97	1,583.10	1,612.01	517.01	598.70	2.28	5,398.92
8	4.38	8.81	5.05	0.62	1.06	0.85	5.31	195.10	0.13	221.31
TOTAL	7,342.95	27,944.48	35,357.16	34,790.03	28.774.13	11,991.15	1,254.60	3,268.22	234.91	150,957.72

Appendix 3.8. (cont'd) Agriculture-Waterfowl Land Capability Cross Tabulation

Southwestern Saskatchewan

By primary CLI agriculture and waterfowl classification (in km^2)

	CLI Agriculture									
Waterfowl Class	1	2	3	4	5	6	7	8	0[a]	Total
1	2.95	692.54	512.97	307.46	580.60	53.12	1.37	0.03	0	2,151.04
2	32.09	1,382.82	4,227.84	1,180.02	2,004.88	85.88	8.62	0	0.03	8,922.18
3	114.37	2,544.97	9,605.96	1,912.62	6,659.98	948.69	5.39	108.60	15.23	21,916.81
4	465.37	6,551.90	19,907.02	6,645.14	6,591.18	830.53	14.37	901.96	69.83	41,977.31
5	256.15	17,414.14	22,570.51	12,215.52	9,019.48	2,729.92	11.45	558.48	212.72	64,888.37
6	2.54	1,234.28	7,393.86	6,380.55	8,381.65	2,468.20	27.84	570.11	288.08	26,747.11
7	9.69	317.45	1,244.98	804.99	4,452.62	8,785.01	16.29	1.97	35.02	15,678.00
8	0	4.07	4.45	0.08	0.52	0.39	0	149.78	0	159.29
TOTAL	883.16	30,142.17	65,467.59	29,446.38	37,690.92	15,912.74	85.31	2,290.93	620.91	182.540.11

Appendix 3.8. (cont'd)

Agriculture-Waterfowl Land Capability Cross Tabulation

Southeastern Saskatchewan

By primary CLI agriculture and waterfowl classification (in km^2)

CLI Agriculture

Waterfowl Class	1	2	3	4	5	6	7	8	0[a]	Total
1	332.58	2,041.55	943.13	309.79	412.43	219.11	0	0	4.64	4,254.23
2	1,315.09	9,844.83	3,365.55	730.71	1,145.58	504.30	0.23	18.65	23.31	16,948.25
3	2,471.41	12,209.46	8,820.19	689.38	2,942.27	545.81	0.62	29.19	111.58	27,819.91
4	3,799.56	11,519.98	15,893.82	1,967.09	1,798.59	547.57	1.37	406.52	498.44	36,432.84
5	2,129.61	4,622.13	5,950.23	3,324.37	2,443.67	704.84	1.14	917.79	864.95	20,958.73
6	659.69	1,241.04	1,316.93	1,171.40	1,003.51	313.83	0.39	5,120.50	418.67	11,245.96
7	74.90	185.60	170.42	92.36	353.35	712.40	0.05	891.71	19.24	2,500.03
8	1.66	0.44	2.18	0	0	0.18	0	24.81	0	29.27
TOTAL	10,784.50	41,665.03	36,453.45	9,285.00	10,099.40	3,548.04	3.80	7,409.17	1,940.83	120,189.32

Appendix 3.8 (cont'd) Agriculture-Waterfowl Land Capability Cross Tabulation

Southwestern Manitoba

By primary CLI agriculture and waterfowl classification (in km^2)

CLI Agriculture

Waterfowl Class	1	2	3	4	5	6	7	8	0[a]	Total
1	0	992.48	5.34	1.71	70.40	4.33	0	0	0	1,074.26
2	0	3,108.78	606.47	101.50	157.00	96.74	138.33	0	106.97	4,315.79
3	52.37	4,559.41	2,190.27	1,265.05	950.55	320.35	297.41	127.38	483.58	10,246.37
4	373.14	7,283.50	6,011.69	3,822.56	1,018.82	602.64	113.49	999.11	657.31	20,882.26
5	1,054.98	8,971.41	6,777.57	10,621.34	3,504.51	812.56	186.01	2,856.34	2,373.69	37,158.41
6	629.50	3,821.52	5,670.79	8,546.68	3,987.72	4,136.98	364.26	2,014.18	9,134.25	38,305.88
7	7.51	300.15	700.90	1,565.80	1,481.16	3,430.33	276.33	514.24	1,232.83	9,509.25
8	8.86	239.03	175.08	0.96	18.34	0.13	0	166.64	0	609.04
TOTAL	2,126.36	29,276.28	22,138.11	25,925.60	11,188.50	9,404.06	1,375.83	6,677.89	13,988.63	122,101.26

[a] Refers to muck soils

Appendix 3.9. Estimated number of dabbling ducks nesting within a number of major crop types or in margin areas associated with those crops, Alberta.

Alberta Census Division	Number of Dabbling Ducks Nesting in Association with:															
	Wheat		Oats		Barley		Rye		Flax		Canola		Tame Hay		Summer Fallow	
	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin
1	29,486	32,762	726	806	2,648	2,942	1,000	1,111	536	596	63	70	25,364	2,818	8,866	29,554
2	18,092	20,102	534	593	7,198	7,998	850	945	407	453	491	546	38,533	4,281	4,076	13,586
3	6,290	6,989	681	757	8,545	9,495	389	432	99	110	230	256	35,268	3,919	1,039	3,463
4	43,049	47,832	4,283	4,759	2,595	2,883	2,148	2,387	73	81	278	309	74,621	8,291	14,789	49,296
5	37,604	41,782	1,599	1,776	17,150	19,056	2,552	2,836	972	1,080	3,201	3,556	34,184	3,798	10,855	36,185
6	4,890	5,433	1,476	1,640	13,690	15,211	889	988	23	26	1,518	1,687	50,930	5,659	1,398	4,658
7	69,271	76,968	11,069	12,299	29,354	32,616	3,540	3,933	818	909	16,239	18,044	120,266	13,363	13,341	44,471
8	1,517	1,686	1,560	1,734	16,293	18,103	71	79	2	2	1,859	2,066	83,431	9,270	615	2,052
10	56,142	62,380	16,762	18,624	50,543	56,159	235	261	335	373	21,683	24,092	133,243	14,805	11,276	37,587
11	3,345	3,716	3,662	4,068	14,239	15,821	19	21	40	44	1,105	1,227	106,177	11,797	1,261	4,204
Total	269,685	299,650	42,351	47,057	162,255	180,283	11,693	12,993	3,305	3,672	46,667	51,852	702,017	78,002	67,517	225,055
Proportion	25.8%		4.1%		15.5%		1.1%		0.3%		4.5%		35.4%		13.3%	

Appendix 3.9 (cont'd) Estimated number of dabbling ducks nesting within a number of major crop types or in margin areas associated with those crops, Saskatchewan.

Saskatchewan Census Division	Number of Dabbling Ducks Nesting in Association with:															
	Wheat		Oats		Barley		Rye		Flax		Canola		Tame Hay		Summer Fallow	
	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin
1	79,146	87,940	8,050	8,944	12,171	13,523	4,911	5,457	3,271	3,635	408	453	84,381	9,376	20,211	67,372
2	75,262	83,624	2,620	2,911	3,546	3,941	594	660	1,135	1,261	34	37	38,149	4,239	19,819	66,064
3	58,270	64,744	1,630	1,811	2,328	2,587	785	872	101	113	0	0	33,484	3,720	18,477	61,590
4	43,834	48,704	1,326	1,473	2,652	2,947	1,014	1,127	41	45	54	60	54,133	6,015	14,773	49,243
5	83,069	92,298	10,190	11,322	14,761	16,401	2,519	2,798	3,324	3,694	2,827	3,141	64,969	7,219	20,760	69,200
6	104,583	116,204	3,571	3,968	10,988	12,209	1,329	1,477	2,630	2,922	480	533	46,861	5,207	28,358	94,526
7	68,476	76,084	2,230	2,477	4,571	5,079	1,570	1,744	416	462	44	49	[illegible],851	5,095	22,439	74,797
8	63,865	70,961	981	1,090	4,689	5,210	2,782	3,091	328	365	11	12	22,661	2,518	20,555	68,518
9	51,467	57,185	5,967	6,630	16,890	18,767	1,112	1,235	585	650	2,170	2,412	67,312	7,479	12,959	43,195
10	73,193	81,325	3,992	4,436	20,096	22,329	1,005	1,116	1,792	1,991	6,537	7,263	34,782	3,865	19,722	65,739
11	69,229	76,921	3,011	3,346	10,333	11,481	1,685	1,872	1,081	1,202	1,873	2,081	65,780	7,309	20,598	68,659
12	50,659	56,288	2,657	2,953	8,216	9,129	2,664	2,960	582	647	1,912	2,125	46,396	5,155	15,772	52,573
13	61,362	68,180	2,884	3,205	12,466	13,852	1,034	1,149	607	675	4,675	5,195	24,946	2,772	18,617	62,058
14	56,193	62,437	2,881	3,201	29,749	33,054	1,129	1,255	4,181	4,645	16,286	18,095	122,739	13,638	11,735	39,118
15	64,833	72,036	3,121	3,468	24,750	27,500	1,115	1,239	1,499	1,666	16,976	18,862	68,305	7,589	14,970	49,899
16	38,935	43,261	5,223	5,803	22,884	25,426	1,760	1,956	422	469	11,085	12,316	84,622	9,402	12,477	41,590
17	30,876	34,306	6,699	7,443	19,814	22,015	1,132	1,257	156	173	17,895	19,884	108,058	12,006	9,087	30,291
Total	1,073,250	1,192,500	67,033	74,481	220,905	245,451	28,140	31,267	22,153	24,615	83,267	92,519	1,013,430	112,603	301,330	1,004,432
Proportion	40.6%		2.5%		8.3%		1.1%		0.8%		3.1%		20.2%		23.4%	

Appendix 3.9 (cont'd) Estimated number of dabbling ducks nesting within a number of major crop types or in margin areas associated with those crops, Manitoba.

Manitoba Census Division	Number of Dabbling Ducks Nesting in Association with:															
	Wheat		Oats		Barley		Rye		Flax		Canola		Tame Hay		Summer Fallow	
	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin	Crop	Margin
3	4,542	5,047	226	251	1,890	2,100	137	153	904	1,005	707	785	3,151	350	84	281
4	5,407	6,008	533	592	4,189	4,654	395	438	2,225	2,473	1,156	1,284	9,618	1,069	382	1,275
5	19,894	22,104	2,391	2,657	8,032	8,924	1,493	1,659	4,767	5,297	2,698	2,998	33,758	3,751	1,812	6,041
6	9,843	19,937	3,065	3,405	4,137	4,597	656	729	1,280	1,423	434	483	30,967	3,441	1,884	6,280
7	3,762	4,180	681	756	2,942	3,268	300	333	673	748	603	670	9,438	1,049	423	1,410
8	2,336	2,595	634	704	1,810	2,011	182	203	507	563	576	640	8,979	998	234	781
9	2,119	2,354	226	251	1,198	1,331	92	102	552	613	307	341	6,003	667	139	462
10	2,854	3,171	136	152	1,162	1,291	5	5	860	955	323	359	2,381	265	86	288
11	520	577	29	32	149	166	3	3	80	89	18	20	433	48	30	101
13	2,202	2,447	167	186	1,038	1,153	19	22	304	338	308	342	7,974	886	243	811
14	2,063	2,292	482	535	1,894	2,104	5	6	291	324	204	227	10,336	1,148	248	827
15	19,855	22,061	2,673	2,970	15,638	17,376	598	664	2,004	2,227	3,929	4,365	41,705	4,634	4,509	15,028
16	4,155	4,617	714	793	2,431	2,701	133	148	94	104	552	613	10,574	1,175	1,074	3,580
17	2,865	3,183	811	901	1,695	1,884	140	156	407	453	602	669	16,358	1,818	609	2,030
18	1,548	1,720	380	422	1,462	1,624	25	28	259	288	323	359	30,898	3,433	407	1,356
20	2,209	2,455	267	297	1,424	1,582	210	233	134	149	338	376	5,530	614	402	1,341
Total	86,175	95,750	13,414	14,904	51,089	56,765	4,394	4,882	15,344	17,048	13,079	14,532	228,105	25,345	12,567	41,890
Proportion	26.2%		4.1%		15.5%		1.3%		4.7%		4.0%		36.5%		7.8%	

Appendix 3.9 (cont'd) Estimated number of diving ducks nesting within a number of major crop types, Alberta.

Alberta Census Division	Number of Diving Ducks Nesting in Association with:								
	Crop or Summer Fallow (total)[a]	Wheat	Oats	Barley	Rye	Flax	Canola	Tame Hay (mostly alfalfa)	Summer Fallow
1	11,715	5,432	134	488	184	99	12	467	4,900
2	9,978	4,135	122	1,645	194	93	112	881	2,795
3	6,554	1,802	195	2,448	111	28	66	1,010	893
4	20,794	8,586	854	518	428	15	55	1,488	8,849
5	24,352	9,244	393	4,216	627	239	787	840	8,006
6	8,886	1,368	413	3,829	249	7	425	1,424	1,173
7	113,506	43,120	6,890	18,273	2,204	509	10,109	7,486	24,914
8	34,146	1,645	1,692	17,666	77	2	2,016	9,046	2,002
10	122,751	33,735	10,669	32,171	149	312	13,801	8,481	21,532
11	41,545	3,775	4,133	16,071	21	45	1,247	11,983	4,270
Total	394,227	114,842	25,495	97,325	4,244	1,250	28,630	43,106	79,334
Proportion	100%	29.1%	6.47%	24.7%	1.08%	0.32%	7.26%	10.9%	20.1%

Appendix 3.9 (cont'd) Estimated number of diving ducks nesting within a number of major crop types, Saskatchewan.

Saskatchewan Census Division	Number of Diving Ducks Nesting in Association with:								
	Crop or Summer Fallow (total)[a]	Wheat	Oats	Barley	Rye	Flax	Canola	Tame Hay (mostly alfalfa)	Summer Fallow
1	41,063	18,359	1,867	2,823	1,139	759	95	1,957	14,065
2	29,182	14,996	522	707	118	226	7	760	11,847
3	24,056	11,500	322	460	155	20	0	661	10,939
4	17,156	7,623	231	461	176	7	9	941	7,707
5	46,014	20,609	2,528	3,662	625	825	701	1,612	15,452
6	49,095	24,067	822	2,529	306	605	110	1,078	19,578
7	29,901	13,722	447	916	315	83	9	919	13,490
8	25,701	12,017	185	882	523	62	2	426	11,603
9	51,573	21,441	2,486	7,036	463	244	904	2,804	16,195
10	60,836	26,308	1,435	7,223	361	644	2,350	1,250	21,266
11	33,269	14,804	644	2,210	360	231	401	1,407	13,214
12	25,033	10,688	561	1,734	562	123	404	979	9,983
13	63,704	27,650	1,300	5,617	466	274	2,107	1,124	25,167
14	73,233	26,062	1,336	13,797	524	1,939	7,553	5,693	16,328
15	65,827	26,017	1,253	9,932	448	602	6,813	2,741	18,022
16	71,329	22,006	2,952	12,934	995	239	6,265	4,783	21,156
17	72,106	19,420	4,213	12,463	712	98	11,256	6,797	17,148
Total	779,080	317,288	23,102	85,386	8,248	6,980	38,985	35,932	263,159
Proportion	100%	40.7%	3.0%	11.0%	1.1%	0.9%	5.0%	4.6%	33.8%

Appendix 3.9 (cont'd) Estimated number of diving ducks nesting within a number of major crop types, Manitoba.

Manitoba Census Division	Number of Diving Ducks Nesting in Association with:								
	Crop or Summer Fallow (total)	Wheat	Oats	Barley	Rye	Flax	Canola	Tame Hay (mostly alfalfa)	Summer Fallow
3	715	362	18	151	11	72	56	25	20
4	8,130	2,745	271	2,127	200	1,130	587	488	582
5	25,010	10,347	1,244	4,177	776	2,479	1,403	1,756	2,828
6	16,292	5,694	1,773	2,393	380	741	251	1,791	3,269
7	6,090	2,051	371	1,603	164	367	329	514	692
8	786	240	65	186	19	52	59	92	72
9	555	213	23	121	9	56	31	60	42
10	473	231	11	94	0	70	26	19	21
11	76	42	2	12	0	7	1	4	7
13	6,040	2,390	182	1,126	21	330	334	865	792
14	761	234	55	214	1	33	23	117	84
15	43,328	14,425	1,942	11,361	434	1,456	2,854	3,030	9,826
16	10,958	3,685	663	2,156	118	83	489	938	2,857
17	11,327	3,250	920	1,923	159	462	683	1,856	2,072
18	14,586	2,718	666	2,566	45	455	568	5,425	2,142
20	13,423	4,676	565	3,014	445	284	715	1,170	2,554
Total	160,550	53,302	8,740	33,224	2,782	8,076	8,411	18,152	27,863
Proportion	100%	33.2%	5.4%	20.7%	1.7%	5.0%	5.2%	11.3%	17.4%

[a] Includes seven major crop types listed as well as summer fallow. Does not include other minor crops.

CHAPTER 4

PESTICIDE USE IN THE PRAIRIES

Examples of how a number of agricultural practices quite unrelated to pesticide use have had a direct impact on waterfowl were given in Chapters 2 and 3. This was necessary to show the close spatial relationship between the two resources and to be able to put any new finding related to pesticide use in the proper perspective. The present chapter addresses pesticide use as a subset of agriculture and attempts to define the spatial and temporal overlap between pesticide use and the waterfowl resource.

4.1 The Compilation of Pesticide Use Statistics

Several methods of documenting pesticide use in the prairies are theoretically available. The most straightforward is an actual census of chemicals sold or used, either at the farm, retail (and hence regional), provincial or national level. When these data are not available or when they cannot be used because they are considered to be confidential, it is often possible to obtain estimates of areas treated for any one pest and then to make some assumptions as to the preferred pesticides. Finally, the least desirable and most imprecise method of estimating pesticide use is by using pest information records and crop data and then making assumptions on both the likelihood of pesticide application and on the identity of the products used. All three of these methods had to be employed to various degrees in this report. The principal data sets are described below. "Private" data compiled either by the pesticide companies or by independent censusers for sale to companies are excluded from the discussion.

4.1.1 Pesticide surveys

Thomson (1974) summarized the available pesticide surveys in Canada. The following account summarizes and updates his findings.

4.1.1.1 Federal statistics

At the federal level, the following sources of information are available:

A. Sales of pest control products by Canadian registrants (Statistics Canada - 1947-77).

This survey operated between 1947 and 1977 and compiled country-wide totals of pesticides sold in Canada. These data were limited in that, by polling registered suppliers only, they excluded direct sales of pesticides by the companies to governments, utilities and other large corporate customers (Thomson 1974). A further limitation is that any data on active ingredients for which there were three or fewer manufacturers were deemed to be confidential and not reported. The result of this policy was that sales statistics were rarely available for single pesticides. Furthermore, only national totals were given which precludes any regional assessments. In any case, these surveys have been discontinued.

B. Information derived from import data (Industry, Trade and Commerce Canada).

Since no pesticides are manufactured in Canada, import records are a logical source of information. They were considered by Thomson (1974) but rejected because of their crudeness. For reporting purposes, pesticides are commonly lumped in large units, often with non-pesticidal commercial commodities. Some pesticides also have non-pesticidal uses. Confidentiality is a problem with these data.

C. Surveys co-ordinated by the Commercial Chemicals Branch of the Environmental Protection Service (Environmental Canada).

Three distinct surveys have recently been established by Environment Canada with the agreement of Agriculture Canada and with, in two of the three cases, the help of Statistics Canada.

In the case of the main survey effort, provincial sales totals are obtained from the registrants for a number of active ingredients. This was done under the joint authority of the "Environmental Contaminants Act" and the "Pest Control Products Act". Under the latter, registrants are obliged to maintain sales records of every registered pesticide for five years after manufacture. Sales data for 24 actives were obtained for the year 1981-82. The survey was increased to cover 89 actives in 1983, 120 actives in 1984 and 195 actives in 1985. As with other surveys of registrants, the data are considered to be confidential and, although they are available to pesticide evaluation personnel of Environment Canada, they cannot be cited in a review exercise such as this one. In 1982, an attempt was also made to "piggy-back" the "Farm Energy Use" survey run country-wide for Agriculture Canada by Statistics Canada. Farmers were asked to list pesticides for which they had spent more than $25 and which had actually been used. Unfortunately, the sampling intensity of this survey was not adequate given the high between-farm variability in pesticide use. Similar attempts to piggy-back the "National Farm Survey" in 1983 were also unsuccessful. Farmers were asked about their use of 15 different active ingredients. Farms were then grouped by watershed for analysis. Again, the results were not considered adequate to be useful to environmental planners or regulators. No references are given for these reports since they are classified as confidential.

D. Information obtained under the "Pest Control Products Act" by Agriculture Canada alone.

As far as we can determine, Agriculture Canada had not, until 1985, availed itself of the wide information gathering powers available to it from the "Pest Control Products Act" with the exception of a small survey of registrants for 31 active ingredients between 1976 and 1978.

These survey results were broken down by broad geographical regions and, in some cases, combined for presentation (e.g. wild oat herbicides or dithiocarbamate fungicides). The results were distributed on a "restricted" basis to federal and provincial regulatory authorities. Agriculture Canada has now served notice that it intends to start collecting sales information under the PCP Act and that these statistics are to be tied to renewal of registration. Unfortunately, only national totals are to be requested. Products covered by the Environment Canada survey (above) will be excluded from this requirement. Again, these data will be strictly confidential and hence not amenable to detailed analysis in an exercise such as this one. The lack of even a simple provincial breakdown of these data will also hamper their meaningful use from an environmental point of view.

4.1.1.2 Alberta

The most extensive information on specific pesticides in use in the prairies is collected by the Alberta government. Since 1970, the province has required all provincially licensed pesticide applicators to report annually on their use of pesticides. Unfortunately, there are indications that the reporting rate is actually in the range of 50-65% (Shaw et al. 1984). The data are not compiled in a form that is readily usable. Also, this survey is very limited for our purpose because it does not include farmers. According to the Alberta Environment Conservation Authority (1976), the survey serves to "assess the performance of the applicator, not the performance of the product or (of) the side-effects, nor is the record directly related to a particular area or parcel of land." This group recommended that a more comprehensive use record be assembled for the province. Small-scale surveys of herbicide use by farmers were carried out by D.A. Drew in 1976-77; the results were summarized by Peden et al. (1982) and are reported here.

4.1.1.3 Saskatchewan

There does not appear to be any comprehensive statistics-gathering exercise in Saskatchewan. However, some of the major insecticide programs (e.g. grasshopper control) have historically been carried out or coordinated by the province and it is possible to obtain some information on these programs from officials in the Department of Agriculture. With the increased availability and choice of insecticides offered to the prairie farmer, the role of the province has gradually decreased over the years so that current information is more difficult to obtain. Small-scale herbicide surveys at the farm level were conducted by A.G. Thomas from 1977-79 and were summarized by Peden et al. (1982).

Starting in 1980, the Environmental Protection Service (Western Region) founded a co-ordinated study of a small (2,893 ha) watershed within the southern Saskatchewan region involving questionnaires to farmers on their use of herbicides as well as sampling of runoff waters from the watershed (Waite et al. 1984). Due to the small area covered by the study, it is not possible to tell if the results are representative of broader farming practices and, therefore, the results are not used in our analyses.

4.1.1.4 Manitoba

Since the late 1960's, Manitoba has coordinated a prairie-wide registrant survey of herbicide use in agriculture. The date are available on an annual basis, but often several active ingredients are combined so as to not divulge market share data which is considered to be proprietary information. Results of these surveys have been summarized and publicized since the late 60's, recently by Daciw (Daciw annual). Information on herbicide use is also collected as part of the Manitoba crop insurance scheme. Farmers are asked about weed problems, herbicide use and performance as well as crop tolerance problems (Todd 1984). Some of this information is gradually being published (e.g. Todd, op. cit.) and more is forthcoming (Appendix 4.1). As far as we can determine, this is the only instance where post-registration performance of pesticides is being systematically monitored by any government agency. A.G. Thomas also surveyed Manitoba farmers in 1978-79 and his data were summarized by Peden et al. (1982). It remains to be determined whether information collected through the Manitoba Crop Insurance Corporation is typical of farmers at large.

4.1.2 Surveys of pest control programs

Information on pest control programs is not collected in any standardized fashion. However, fragmentary information is contained in a publication of Agriculture Canada (annual): The Canadian Agricultural Insect Pest Review (formerly "Canadian Insect Pest Review"). In these reports, regional summaries of pest problems are provided by the regional agriculture research staff. Information on any given control program may or may not be included in the report depending on the involvement of the Department of Agriculture in the control program, the magnitude of the program and the pest itself. Emphasis appears to be on new pests or new control programs and information is often lacking on well established pests which demand yearly regular treatments. Since this is the only published source, information on pest levels is spotty and difficult to tally in any uniform fashion.

4.2 Insecticide Use and Overall Market Share

As discussed in Section 4.1, it is very difficult to get an accurate estimate of the identity and quantity of insecticides used in agriculture. When registrant surveys are available, the information has to be treated as confidential which greatly reduces the value of such surveys to regulators and research planners. Furthermore, even when available, sales totals are inadequate for the purpose of hazard assessment. One must still estimate the frequency, method and timing of application as well as the exact areas treated since all of these factors have a bearing on the environmental hazards associated with pesticide application. Appendix 4.2 summarizes the use patterns of the main insecticides employed on prairie crops. We also indicate whether the provincial agencies have provided economic thresholds to help the user in his decision-making.

Following the 1984 and 1985 spray seasons, insect pest control specialists with the provincial departments of agriculture were asked to rank insecticides by their frequency of use in their respective provinces (Alberta - M. Dolinsky, Saskatchewan - L. Harris, Manitoba - A. Kolack and D. Smith, pers. comm.). Following the 1984 season, they all agreed that carbofuran was by far the most heavily used insecticide on the prairies. Use included both foliar and granular use, the latter especially for flea-beetle control in canola. It was their perception that the other most commonly used insecticides were chlorpyrifos, malathion and dimethoate followed by methomyl and carbaryl. These respondants mentioned the recent declining popularity of dimethoate and, interestingly, none of the three considered that there was much use of the registered synthetic pyrethoids. Initial trials with cypermethrin for grasshopper control had been disappointing and were thought to have affected pyrethroid use in general. However, the 1983 and 1984 confidential EPS registrant survey indicated that at least one pyrethroid (deltamethrin) had in fact acquired a substantial part of the insecticide market; confidentiality requirements prevent us from disclosing exact figures. Although relatively small quantities had been sold, they accounted for a large use on the basis of area treated as a result of the very low application rates.

The growing popularity of deltamethrin became clear after the 1985 spray season. Its use in Saskatchewan is now thought to be second only to carbofuran (M. Dolinsky, pers. comm.). Shortages in supplies of carbofuran encouraged farmers to switch to deltamethrin and, since its efficacy was good, the product may have gained a wide following. During the 1985 grasshopper outbreak, carbofuran still maintained the largest share of the market with deltamethrin and carbaryl competing for second place (See Section 4.2.1 and Table 4.2). The province of Alberta should have good records from this particular outbreak as a result of a provincial rebate program on all purchases of insecticides.

Our pundits made several predictions. As a result of recent efficacy tests, the spray dosage of chlorpyrifos will be halved for grasshopper use, making this product more competitive. Also, there may be a slight revival of dimethoate in the form of bran bait although the objectionable odour of this product puts it at a major disadvantage. Non-registered (and therefore illegal) use of other insecticides on bran bait is likely and indeed, such use has been openly encouraged by the Research Division of Agriculture Canada in the form of a press release to farmers (Ewen & Mukerji 1985). Finally, it is acknowledged that farmers are increasingly resorting to mixing their own "cocktails" of insecticides. In view of potential synergism (e.g. between organophosphates and pyrethroids) this trend is worrisome.

For the present, however, the reduction in dimethoate use, the gaining popularity of at least one synthetic pyrethroid and the continuing reliance on carbofuran are all factors which will have a bearing on our hazard assessment.

4.2.1 Grasshopper control

Of the economically important insect pests of prairie crops, grasshoppers are potentially the most important because they occasionally give rise to outbreaks that cover areas greater than those covered by any other insect pest species. Because of the wide fluctuation in their year-to-year importance, they are the only pests for which infestation predictions are annually compiled prairie-wide. Since the 1930's, surveys have been run to identify regions where large infestations might appear (Gage & Mukerji 1977). Historically, much of the pesticide use against grasshopper outbreaks has been under provincial control. The provinces were either directly involved with the spraying or they co-ordinated the distribution of insecticides to municipalities or individual farmers. For this reason, records of insecticide use against grasshoppers are better than those for other pest species although, even here, it is difficult to piece together a complete picture of the total use. As mentioned earlier, there is now less involvment on the part of provincial governments which makes information-gathering more difficult.

The Alberta Environment Conservation Authority (AECA 1974a) documented the use in Alberta of malathion, dieldrin and dimethoate, and summarized the infestation forecasts for the years 1960-72 (Table 4.1). Infestation forecasts were rated as "light", "moderate" or "severe" and in later years a "very severe" rating was added. The "land infested" figure was obtained directly from the forecasts. The "crop menaced" figure was probably estimated on the basis of the percentage of lands devoted to crops and the "crop sprayed" figure was determined by assuming that most of the spraying took place on croplands at the minimum recommended spray rate. In fact, most grasshopper spraying takes place in non-crop areas in close proximity to crop areas: along fencelines, roads and slough margins. During an outbreak, it is customary for these non-crop areas to be sprayed from two to seven times whereas crop areas and pastures are usually sprayed once and occasionally twice. The most striking observation to be made from Table 4.1 is the wide variation in spray area from one year to the next (from 0.7% to 61% of the menaced crop area) which reflects the nature of the pest and the difficulty of predicting spray activity. The year 1962 is acknowledged to be one of the worst outbreak years in recent recorded history, although it is now likely to have been superseded by 1985. Those two years are therefore useful as worst-case estimates of the control measures that may take place during heavy infestations. The total area sprayed in Alberta in 1962 was just over ½ million ha or 55% of the crop area rated with a moderate-severe infestation. In 1985, it is estimated that enough insecticide was used to cover .7 million ha (Table 4.2).

It is also clear from Table 4.1 that, following the removal of dieldrin, dimethoate became the insecticide of choice for grasshopper control in Alberta. This was also the case in Saskatchewan (Table 4.3) where dimethoate has, for a number of years, been bought in bulk by the Saskatchewan Agriculture Department and distributed at cost to farmers. Alberta had a similar program which was phased out in 1980. As discussed earlier, there has recently been a sharp reduction in the use of dimethoate for grasshopper control in favor of other registered alternatives. It has

Table 4.1. Insecticides used for grasshopper control (1961-72) in Alberta[a].

	Area in ha (thousands)				Insecticides Used[d] (kg)		
Year	Land Infested[b]	Crop Menaced[c]	Crop Sprayed	% of Menaced Crop Sprayed	Malathion	Dieldrin	Dimethoate
1960	2,752.0	364.2	66.0	18.1	f	f	f
1961[e]	4,856.4	680.0	412.6	60.6		28,890	
1962	6,272.8	1,011.7	142.2		79,675		
			413.0			28,917	
			555.2	54.9			
1963	7,972.6	1,315.3	23.7		16,330		
			259.0			18,144	
			112.6				23,678
			395.3	30.0			
1964	6,475.2	907.2	0.8		657.7		
			65.0			4,554	
			11.8				2,496
			77.6	8.5			
1965	4,370.8	526.1	0.2		113.4		
			6.9			480.8	
			1.2				245
			8.3	1.6			
1966	1,740.2	323.8	0.2		181.4		
			2.5			177	
			1.5				313
			4.2	1.3			
1967	2,144.9	364.2	4.5			317.5	
			4.4				936
			8.9	2.4			

Table 4.1 (cont'd)

Year	Area in ha (thousands)			% of Menaced Crop Sprayed	Insecticides Used[d](kg)		
	Land Infested[b]	Crop Menaced[c]	Crop Sprayed		Malathion	Dieldrin	Dimethoate
1968	2,307.8	323.8	7.2	2.2			1,544
1969	1,902.1	283.2	3.0	1.0			639
1970	440.3	80.9	0.6	0.7			131
1971	1,074.9	161.9	1.9	1.1			338.3
1972	4,058.7	?	22.9	-			4,884
				Average = 15.2 s.d. ±21.8			

[a] Source: Modified from Alberta Environment Conservation Authority 1974a and the Canadian Agriculture Insect Pest Review 1963

[b] All the land in areas where grasshopper populations are likely to cause economic damage (light, moderate and severe); includes rangeland, rough land and seeded acreage (based on grasshopper densities the previous year)

[c] The acreage of land seeded to crops in the infested area where the grasshopper density was in the moderate to severe categories the previous year (i.e. where the densities were such that the crop would be damaged)

[d] Active ingredient

[e] 27 kg of Aldrin also used

[f] Insecticide not specified

Table 4.2. Information on the 1985 grasshopper spray program[a].

Province	Estimate of Area Sprayed at Least Once ($\times 10^6$ ha)	Extended Spray Area ($\times 10^6$ ha) Based on Insecticide Sales	Rough Estimate of the Proportion of Aerial Spraying	Main Insecticide Used (estimated % of market share)
Saskatchewan	2.0-2.8	2.8-3.6	< 10%	carbofuran 40% deltamethrin 25% carbaryl 25%
Alberta	.53[b]	.70	< 20%	carbofuran 66% deltamethrin 16% carbaryl 16%
Manitoba	.18-.30[b]	.24-.40	< 25%	carbofuran 40% deltamethrin 10% carbaryl 50%
Prairie total	2.7-3.6	3.7-4.7	< 13%[c]	carbofuran 44%[d] deltamethrin 22%[d] carbaryl 26%[d]

[a] Sources: M. Dolinski (Alberta), L. Harris (Saskatchewan), D. Smith (Manitoba), pers. comm.
[b] Calcuiated using the same proportion of repeat spraying as Saskatchewan
[c] Weighted average using maximum estimated spray areas - see Chapter 5 for discussion
[d] Weighted average using maximum estimated spray areas.

Table 4.3. Area sprayed with dimethoate for grasshopper control in Saskatchewan[a].

Year	Quantity of Active[b] (in kg)	Area Sprayed[c] (in ha)	% of Total Land Sprayed in Saskatchewan for Grasshopper Control[d]
1962[e]	809,400	2,023,500	100
1971	1,802	4,506	N.D.[f]
1972	36,377	10,944	N.D.
1973	40,979	102,449	N.D.
1974	158,637	396,593	43.5
1975	48,824	122,060	60.3
1976	13,092	32,731	N.D.
1977	0	0	0
1978	0	0	0
1979	9,547	23,866	N.D.
1980	38,777	96,943	47.9
1981	20,002	50,006	24.7
1982	55,483	138,707	11.4
1983	12,432	31,080	7.8

[a] Source: L. Harris, Saskatchewan Agriculture, pers. comm.

[b] Quantities were originally expressed as gallons or litres of formulated product and were converted to kg with a knowledge of the guarantee for each formulation (1962-79 = Cygon 480 EC, 480 g ai/l; 1980-83 = Cygon Hopper Spray 400 EC, 400 g ai/l)

[c] Calculated assuming application rate of 400 g ai/ha once

[d] Total area sprayed from Table 4.5 except for 1962 value when dimethoate was the only product used

[e] 1962 quantity back-calculated from an application rate of 400 g ai/ha

[f] N.D. - Not Determinable.

been suggested that rebate programs can also affect insecticide choice (L. Harris, pers. comm.). Farmers are more likely to apply more costly materials if they are subsidized by the public purse since cost may often be (rightly or wrongly) equated with quality and efficacy.

The total land area (cropland as well as pasture rangeland) placed in the moderate-severe category for the years 1973-83 in all prairie provinces is given in Table 4.4. Of note is the large relative importance of Saskatchewan as a potential centre of infestation. An area of southern Saskatchewan (part of the "Palliser Triangle") has traditionally had the most severe outbreaks. The actual areas sprayed are more difficult to determine. Table 4.5 summarizes the information published in the "Canadian Agricultural Insect Pest Review" (see 4.1.2 for a discussion of this information base) on the total area sprayed for grasshopper control and compares the areas sprayed to the infestation predictions. Once again, it is obvious that the infestation predictions are not very useful in predicting spray area. The proportion of "menaced" land that was actually sprayed varies by three orders of magnitude in the examples given (0.01% to 18%). Both in 1974 and 1982, the total area sprayed in Saskatchewan was in the vicinity of one million ha. For comparison, the area sprayed for grasshoppers in Saskatchewan in the outbreak of 1962 was two million ha and, in 1985, an estimated 2.0-2.8 million ha were sprayed at least once; 2.8-3.6 million ha is the "extended" estimate (including re-application) (Table 4.2). Forecasts in the early summer of 1985 predicted that over five million ha would be sprayed but, as the season progressed, many farmers abandoned spraying altogether. In Alberta, also, the dry conditions which prevailed through the summer discouraged farmers from trying to protect the crop from the grasshoppers. In contrast, during the 1974 and 1982 heavy-spray years, the numbers of "July ponds" in southern Saskatchewan were almost at a record high (Section 3.4.3) indicating very wet conditions, at least along the surveyed transects. The total (extended) prairie area treated for grasshoppers in 1962 was just under three million ha; it was possibly as high as 4.7 million ha in 1985 (Table 4.2).

The use of insecticides for grasshopper control can be summarized as follows:

A. The area sprayed in any one year is extremely variable. Based on the last decade (assuming that 10% of the total land with moderate-severe infestation prediction will be sprayed), we expect two million hectares to be sprayed for grasshoppers in a moderate-serious outbreak year. Most of the spraying would take place in Saskatchewan and, based on the usual severity of the outbreak, probably in the Palliser Triangle. However, we note that the total prairie area sprayed for grasshoppers was as high as three million hectares in 1962 and may have been almost five million hectares in 1985.

B. The severity of the outbreak may be accurately predicted (otherwise, surveys would presumably not continue) but the actual magnitude of spray cannot be forecasted from the current methods of estimating infestations.

Table 4.4. Grasshopper forecasts for the prairie provinces 1973-1983 (excluding light infestation category)[a].

	Total Area[b] (ha)			
Year	Alberta	Saskatchewan	Manitoba	Prairies
1973	2,706,539	4,829,813	718,203	8,254,556
1974	2,330 991	9,146,808	524,473	12,002,272
1975	3,202,174	10,498,783	697,224	14,398,181
1976	2,427,866	11,178,440	228,474	13,887,580
1977	683,652	7,440,293	0	8,120,945
1978	123,802	2,954,154	0	3,077,956
1979	216,524	4,352,754	68,477	4,637,755
1980	45,500	12,706,540	159,544	12,911,548
1981	999,900	9,541,200	89,300	10,630 400
1982	1,480,900	12,354,300	47,900	13,883,100
1983	3,086,700	17,808,000	50,500	20,961,100

[a] Source: Modified from The Canadian Agricultural Insect Pest Review (Agriculture Canada) unless otherwise specified

[b] Land areas that would potentially require chemical control and includes pasture rangeland, rough land and seeded acreage.

Table 4.5. Areas of land sprayed for grasshopper control as a percentage of forecast from previous year[a].

Province	Year	Area in ha: Land Potentially Requiring Control[b]	Area in ha: Land Sprayed[c]	% of Land Potentially Requiring Control that was Sprayed
Manitoba	1971	293,187	2,145	0.73
Saskatchewan	1974	9,116,808	910,575	10
	1975	10,498,783	202,350	2.0
	1977	7,440,293	809	0.01
	1980	12,706,540	202,350	2.0
	1981	9,541,200	202,000	2.0
	1982	12,354,300	1,214,000[d]	9.8
	1983	20,012,334	400,000[d]	2.0
	1984	12,680,700	607,050	4.8
	1985	20,163,900	3,600,000[d]	18
Alberta	1981	1,480,900	6,913	0.47
	1983	3,102,600	35,000	1.1

[a] Source: Modified from The Canadian Agricultural Insect Pest Review (Agriculture Canada)

[b] Area of land rated as moderate, severe and very severe

[c] Estimates for all insecticides used for grasshopper control, quantities of specific actives not recorded (L. Harris, Saskatchewan Agriculture, pers. comm.)

[d] Highest estimate.

C. There is a general trend away from dimethoate to other registered alternatives. This trend is potentially very important in estimating the hazard of insecticides to the waterfowl resource (Chapter 6).

D. The soil moisture conditions during years of heavy spraying are unclear. Although there is evidence that outbreaks are exacerbated by dry conditions, spraying may actually be heavier under wet conditions when there is a better expectation of harvesting a crop (see Section 4.2.4.2).

4.2.2 Other insect control programs and total insecticide use

Table 4.6 summarizes the information obtained from the Canadian Agricultural Insect Pest Review between 1970 and 1983 for pests other than grasshoppers. The shortfalls and general inadequacy of this data base were considered earlier. It is noteworthy that most of the pests for which information is available are pests of canola; canola was gaining in popularity during these years which may explain this emphasis. In contrast with the few million hectares occasionally sprayed for grasshopper control, the most extensive spray program recorded for any of the other insect pests is for the bertha armyworm in 1972 when a reported 270,000 ha were treated prairie-wide. In fact, this figure is lower than that given by the three provincial agricultural departments to the press at the time (Table 4.7). According to those provincial sources, nearly 400,000 ha of canola were treated for bertha armyworm in 1971 and 317,000 ha in 1972. Estimates of the infestation ranged from ½-1 million ha. Lannate^R (methomyl) was the main insecticide applied and most of the crops were sprayed by air in this massive program.

In addition to the bertha armyworm spray program, in 1985 an estimated .11 million ha were treated for the pale western cutworm in Alberta alone (M. Dolinski, pers. comm.). In the same year, an estimated .1-.2 million ha were sprayed for the orange blossom wheat midge (Sitidiplosis mosellana) (L. Harris, pers. comm.), which has recently been elevated to the status of major pest following a series of outbreaks since 1982 in northeastern Saskatachewan. This pest, in contrast to grasshoppers, appears to favor areas characterized by high soil moisture. Finally, in August 1985, an emergency spray program was requested for a control program of 100,000 ha for the diamond-back moth in canola. Provincial pest control experts estimate that, in fact, 450,000 ha and 30,000 ha were sprayed for this pest in Saskatchewan and Alberta, respectively (L. Harris & M. Dolinski, pers. comm.). Whereas most cutworm control is achieved through ground delivery of the insecticide, virtually all the treatments for wheat midge and diamond-back moth are aerially applied.

A further failure of "Agricultural Insect Pest Reviews" is that they cannot provide a complete picture of "background" insecticide use in average years when no one pest in particular is in an outbreak situation. Two sources of information are available. The first is the Census of Agriculture which provides information on pesticide use every decade. Farmers are asked to report the total "land or crop area sprayed or dusted for controls of a) weeds and brush, and b) insects and disease." Census values are available for the years 1970 and 1980. In Table 4.8 these values

Table 4.6. Summary of reported estimates of crop sprayed for pest control (other than grasshopper), 1970-1983, from Canadian Agricultural Insect Pest Review (Agriculture Canada).

		Reported Area Sprayed[a] (in ha)		
Pest and Crop	Year	Alberta	Saskatchewan	Manitoba
Bertha armyworm (Mamestra configurata) on canola	1971	N.R.[b]	c	6,200
	1972	89,034[d]	161,880	20,235
	1973	22,258	809	N.R.
	1979	1,214-2,023	N.R.	N.R.
	1980	6,000	40,470	2,832
	1981	2,700	12,141	5,680
	1983	1,480	0	0
Beet webworm (Loxostege sticticalis) on canola	1971	16,200[d]	c	c
Flea beetle (Phyllotreta striolata) on canola	1976	N.R.	c	N.R.
	1978	2,995	N.R.	N.R.
	1983	N.R.	>100,000	N.R.
Clover cutworm (Scotogramma trifolii) on canola	1971	455	N.R.	N.R.
	1972	455	N.R.	N.R.
	1982	10,000	N.R.	N.R.
	1983	300	N.R.	N.R.
Cutworms (pale western, red-backed, dark-sided, early) (Agrotis orthogonia) (Euxoa sp.) on cereals	1983	11,000	18,000	N.R.
Alfalfa looper (Autographa californica) on canola	1973	6,070	N.R.	N.R.
	1978	1,336	N.R.	N.R.
Diamond-back moth (Plutella xylostella) on canola	1977	N.R.	84,987	N.R.
	1978	405	N.R.	N.R.
	1983	N.R.	2,000	N.R.
Alfalfa weevil (Hypera postica) on alfalfa	1979	1,214	N.R.	N.R.
Sweetclover weevil (Sitona cylindricollis) on clover	1983	N.R.	1,000	N.R.

Table 4.6 (cont'd)

Pest and Crop	Year	Reported Area Sprayed[a] (in ha)		
		Alberta	Saskatchewan	Manitoba
Lygus spp. on alfalfa	1981	370	N.R.	N.R.
	1982	360	N.R.	N.R.

[a] For cereal, oil-seed and forage crops in addition to rangeland (foliar treatments only)

[b] N.R. - Area sprayed (if any) not reported

[c] Expressed as a % of crop infested or stated as extensive

[d] Largest infestation recorded.

Table 4.7. Estimated average area (ha) treated during the 1971-72 bertha armyworm infestation[a].

	Alberta	Saskatchewan	Manitoba	Total
1971	83,000	303,500	4,000	390,500
1972	89,000	207,600	20,200	316,800

[a] Sources:
1. The Province, Innisfail, Alberta - 12 September 1972
2. The Sun, Swift Current, Saskatchewan - 7 September 1972
3. The Review, Coronation, Alberta - 9 September 1971
4. The Free Press, Winnipeg, Manitoba - 28 August 1972.

Table 4.8. Reported areas (in thousands of ha) sprayed or dusted for insect control in the prairie provinces, 1962, 1970-82.

Year	Alberta		Saskatchewan		Manitoba		Prairies	
	Provincial estimates	Census estimate	Provincial estimates	Census estimate	Provincial estimates	Census estimate	Provincial estimates	Census estimate
1962	555.2[a]	--	2,023.5[a]	--	349.8[a]	--	2,928.5	--
1970	0.6[a]	161.7	--	244.3		126.6	0.6	532.6
1971	18.5[b]		4.5[a]		8.3[b]		31.3	
1972	112.4[b]	--	172.8	--	20.2	--	305.4	--
1973	28.3	--	103.2[b]	--	--	--	131.5	--
1974	--	--	910.6[a]	--	--	--	910.6	--
1975	--	--	202.3[a]	--	--	--	202.3	--
1976	--	--	32.7[a]	--	--	--	32.7	--
1977	--	--	85.8[b]	--	--	--	85.8	--
1978	4.7	--	--	--	--	--	4.7	--
1979	3.2[c]	--	23.8[a]	--	--	--	27.0	--
1980	6.0	210.2	242.8[b]	608.2	2.8	259.8	251.6	1,078.2
1981	10.0[b]	--	214.1[b]	--	5.7	--	229.8	--
1982	10.3	--	1,214.0[a]	--	--	--	1,224.3	--

[a] Grasshopper spray only (from Table 4.1, 4.3, 4.5 and The Canadian Agricultural Insect Pest Review)

[b] Grasshopper spray area totals included in estimates

[c] Calculated using highest estimate from Table 4.6

are compared to the estimates obtained by summing all the insect control programs reported in the Agricultural Insect Pest Review. There is little correspondence between the two estimates and we must assume that the census estimate is a better reflection of actual insecticide use for the reasons outlined above. (Note: Fungicide use on cereal and forage crops is limited and we assume that the total area treated for insects and disease represents insecticide use only.) Areas of chemicals "sprayed or dusted" would not include seed treatments and probably not any applications of granulars. Although the intent of the questions was to establish the area "sprayed" or "dusted" at least once, it is possible that some of the respondants multiplied the treated land area by a certain factor to account for re-application (Rick Burroughs, Statistics Canada, pers. comm.). The extent to which this happened is not known. One of the most useful facets of the agricultural censuses is that the proportion of cropland treated with insecticides is available by census division and this can be incorporated into hazard scenarios (Chapter 8). Although these proportions must vary greatly between years, such a division allows for a realistic estimate of the number of birds at risk in a given year as a result of normal insecticide use.

A final check on total insecticide use is from the confidential registrant survey conducted by the Environmental Protection Service of Environment Canada (Table 4.9). Based on the sales volumes of the 12 major insecticides in use in the prairies, we can estimate the total area treated assuming a one-time application at recommended label rates for major prairie pests. The main bias here is overestimation of the area treated as a result of multiple applications. This bias is likely to be more severe in Manitoba as a result of the large relative importance of vegetable crops which, although restricted to small areas, tend to be very insecticide-intensive. A further problem with the Manitoba estimate is our inability to separate the agricultural use of malathion from its use in the extensive spray program to control mosquito vectors of equine encephalitis as well as its use on stored grain and for the abatement of the forest tent caterpillar. The large mosquito adulticiding program takes place primarily in urban areas and presumably does not have much relevance to wetland contamination. The larviciding programs, however, involve a direct treatment of waterbodies, primarily with chlorpyrifos, methoprene and, recently, the biological Bacillus thuringiensis H-14. These applications are outside the scope of this report. Keeping these known biases in mind, it is interesting that the total calculated prairie area sprayed with insecticides in 1983 (918.4-1,771.6 thousand ha) agrees with the 1980 census estimate of 1,078.2 thousand ha. Neither year had major grasshopper outbreaks but there was a moderate infestation of bertha armyworm in rape-seed crops in 1980.

In summary, at the present time, normal (average) areas sprayed or dusted (note: crops are rarely dusted nowadays) annually with insecticides are in the range of 1-1.5 million ha for the prairies, although it should be noted that the 1970 census figure was only .5 million ha. As discussed above, another two million ha may be added to this estimate for a total of 3-3.5 million ha during years of serious grasshopper outbreaks. Should there be a coincidental infestation of bertha armyworm or diamond-back moth, a further .4-.5 million ha could be added to the total spray area for either pest. Were we to assume that most of this insecticide use takes place on

Table 4.9. Estimated area sprayed in 1983 for insect control based on the EPS registrant survey (see text).

Insecticide	Application Rates used to Calculate Area Sprayed (g/ha)	Estimated Area of Application		
		Alberta	Saskatchewan	Manitoba
		(X 1000 ha)	(X 1000 ha)	(X 1000 ha)
Azinphos-Methyl	150 - 1000	a	a	a
Carbaryl	1100 - 2250	a	a	a
Carbofuran	150 - 275	a	a	a
Chlorpyrifos	350 - 550	a	a	a
Deltamethrin	5 - 10	a	a	a
Diazinon	550	a	a	a
Dimethoate	200 - 550	a	a	a
Malathion	425 - 850	a	a	a
Methidathion	175 - 550	a	a	a
Methomyl	150 - 500	a	a	a
Methoxychlor	1100 - 2750	a	a	a
Trichlorfon	275 - 1100	a	a	a
All Insecticides	Not applicable	133.4-248.3	307.3-572.2	477.7-951.1[b]

[a] Data cannot be released due to confidentiality requirements

[b] See text for explanation of this provincial total

improved lands, estimated at 37.7 million ha in 1981 (Statistics Canada - 1981 Census of Canada), insect control programs would cover 3-4% of improved lands during normal years, 8-9% during serious grasshopper outbreaks and 10-11% if a grasshopper outbreak was to coincide with another serious outbreak such as by bertha armyworm or diamond-back moth. In 1985, definitely a year to be remembered, we estimate that 6.5-7 million ha were sprayed with insecticides, an area which corresponds to 17-19% of all improved lands or roughly 21-23% of all cropland (land classified under CLI agriculture Classes 1-3 (Section 3.3)) in the Canadian prairies.

4.2.3 Temporal overlap between insecticide use and duck breeding

Table 4.10 summarizes the timing information available for some of the major prairie crop pests. The temporal overlap between pesticide application and the duck breeding season (Chapter 2) is extensive. If we assume that the time between hatch and fledging is the most sensitive to impact from insecticide sprays (Chapter 2), then late May-mid July is the critical period for most duck species. On that basis, some pest control programs are more likely to cause problems than others. For example, an insecticide foliar spray for flea beetle control (late May-June) has the potential to have more impact on the bulk of nesting duck species than a similar spray for bertha armyworm (August). On the other hand, it could be argued that late sprays have the potential to have more impact on the less numerically important species such as diving ducks and that these species require greater protection as a result of their small population sizes and slower "rebound" capability (Cooch 1981). There does not, therefore, appear to be any way of reducing the temporal overlap between waterfowl nesting and insecticide use.

4.2.4 Factors affecting insecticide use

4.2.4.1 Proximate factors

The main factors, besides crop type, which influence insect pest levels are: climate and geography, the extent of the crop (both spatially and temporally, e.g. crop rotation), crop sanitation and the developmental stage of the crop (modified from DeWitt 1983). Factors which, in turn, affect pesticide use are economic threshold levels, as well as peer and institutionalized pressure to spray. It is clearly beyond the scope of this report to review the factors which control every agricultural spray program in the prairies. Instead, we will restrict our discussion to the program which may result in the greatest amount of insecticide use: grasshopper control. It should be pointed out that not all pests respond similarly to the same factors. What may favor one particular pest may prevent the outbreak of another.

Table 4.10. Period of insecticide application for several common insect pests in the prairies[a]. Solid line refers to peak spray period.

Crop	Pest
Cereals	grasshoppers (*Melanoplus*, *Camnola*)
	armyworm (*Pseudaletia unipuncta*)
	cutworms (*Agrotis*, *Euxoa*)
	aphids (*Macrosiphus*, *Rhopalosiphum*)
	orange blossom wheat midge (*Sitidiplosis mosellana*)
Canola	flea beetle (foliage spray) (*Phyllotata*)
	red turnip beetle (*Entomoscelis americana*)
	bertha armyworm (*Mamestra configurata*)
	diamond-back moth (*Plutella xylostella*)
	clover cutworm (*Scotogramma trifolii*)
	cutworms (*Agrotis*, *Euxoa*)

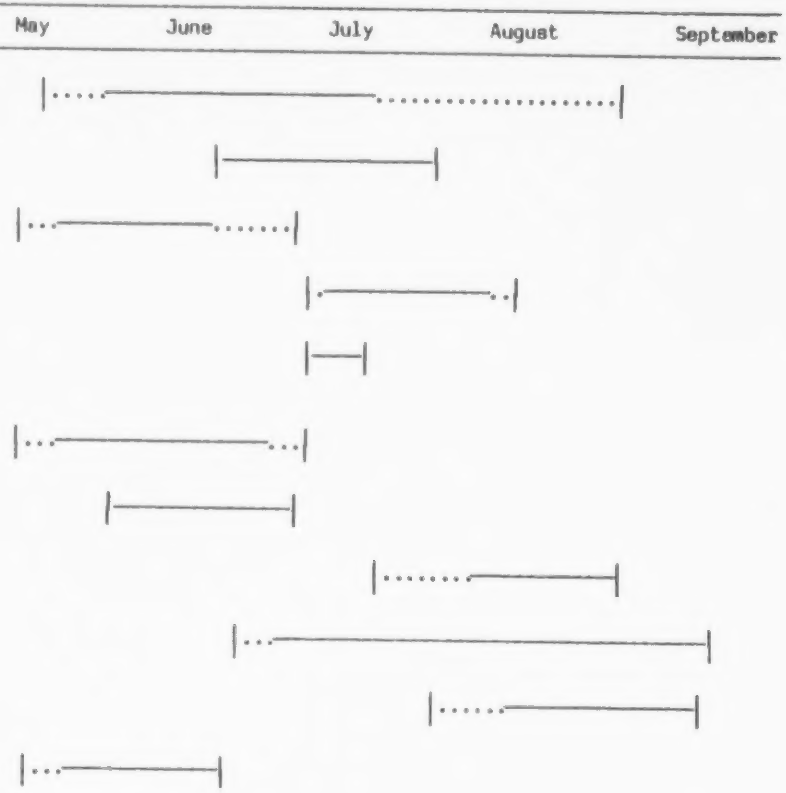
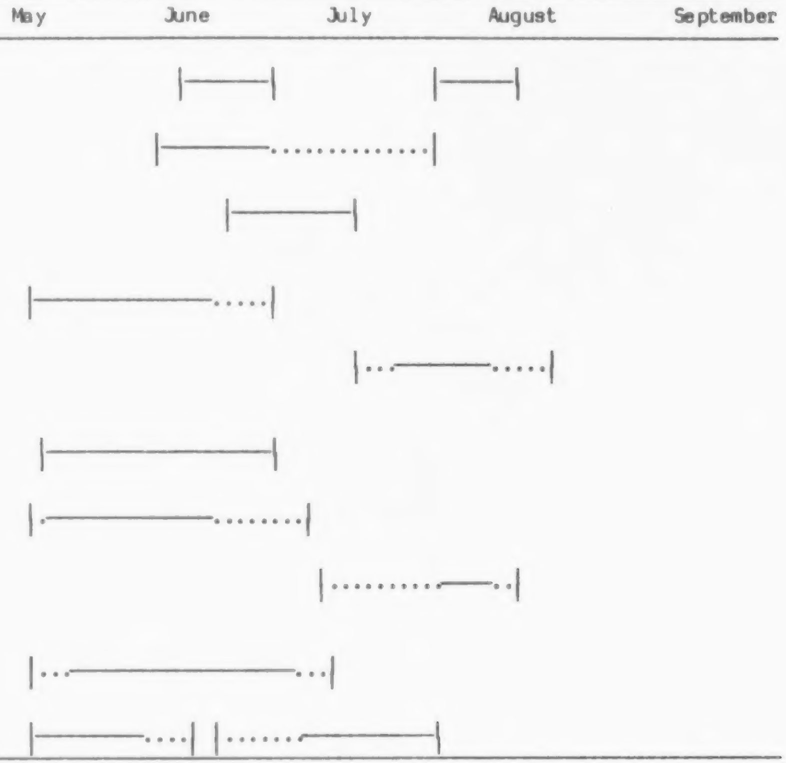


Table 4.10 (cont'd)

Crop	Pest
Alfalfa	lygus bugs (*Lygus*)
	alfalfa plant bugs (*Adelphocoris lineolatus*)
	alfalfa looper (*Autographa californica*)
Clover	sweet clover weevil (*Sitona cylindricollis*)
	European cornborer (*Ostrinia nubilalis*)
Flax	grasshoppers (*Melanoplus*, *Camnola*)
	cutworms (*Agrotis*, *Euxoa*)
	aphids (*Rhopalosiphum*, *Macrosiphum*)
Sunflower	cutworms (*Agrotis*, *Euxoa*)
	sunflower beetle (*Zygogranna exclantionis*)



---

[a] Sources: Canadian Agricultural Insect Pest Review (annual). Personal communications from A. Kolack, D.L. Smith (Manitoba); H. Phillip, M.G. Dolinski (Alberta).

4.2.4.2 Climate, geography and grasshoppers

We saw in 4.1.3 that insecticide use against grasshoppers was both erratic and unpredictable from one year to the next. From a proximate point of view, climate is the most important factor affecting the size of the grasshopper infestation which, in turn, dictates the level of insecticide use. A complicating factor is the direct effect of climate on the crop which is often difficult to separate from losses due to pests. For example, much of the grasshopper damage in the prairies occurs when crop yield has already been depressed as a result of several years of drought (Gage & Mukerji 1978). Similarly, much of the real grasshopper damage (as opposed to the infestation level or predicted infestation level) takes place in areas which can be predicted on the basis of the heat to precipitation ratio or overall water stress. In Saskatchewan, these areas tend to be in the southwest area of the province. Major grasshopper outbreaks usually follow one or more years of drought which allows pest numbers to increase beyond spray threshold levels. The year in which spraying itself takes place is not necessarily a dry year as exemplified by the years 1974, 1982 and 1985 which all received heavy sprays but had very favorable May pond counts (Section 3.4.3). However, a lack of precipitation in spring and summer, as occurred in 1985, may exacerbate the problem and trigger a severe infestation. Although a large amount of spraying took place in 1985, there would have been more spraying had there been more summer rains and a greater hope of harvesting a crop (L. Harris, pers. comm.).

4.2.4.3 Ultimate factors, cost benefit of insecticide use

In the case of grasshopper control, farmers are commonly placed in a position of using insecticides to minimize their losses in yield rather than to maximize their profits since the total cost of protection is often added to the total monetary loss incurred (Gage & Mukerji 1978). One figure commonly cited for the U.S. is that, on average, \$1 spent on insecticides creates a net return of \$3-5 (Metcalf 1980). However, it is acknowledged that for some crops, the benefits are much greater and that the ratio of 3-5:1 is biased as a result of the low rates of return of insecticides on large field crops such as wheat or tobacco (Stemeroff & George 1983). It is also necessary to clarify that the cost side of the equation, which is steadily increasing, only includes the direct, tangible costs associated with crop loss or crop protection. According to Metcalf (1980), wholesale values for well-established insecticides rose by 21.7% per year over the period 1970-77. Reasons for this cost increase have been the increased costs of petro-chemicals, the switch to less residual insecticides, the increasing complexity of insecticide molecules requiring more extensive screening and higher production costs and the increased registration costs. The costs of synthesizing a pyrethroid insecticide is over 100 times that of synthesizing DDT. The market price of most of the insecticide-intensive crops over the same period has been relatively static.

Another reason for escalating costs of insecticides has been the failure to implement Integrated Pest Management (IPM) on a large scale resulting in the decreased usefulness of some very good insecticides as a result of pest resistance, the appearance of secondary pests, decimation of natural checks such as parasites and predators, etc.

Also not included in the cost side of the equation are such intangibles as environmental contamination and loss of wildlife or wildlife habitat, human and livestock poisonings and loss of honeybees (Metcalf 1980). The benefits of insecticide use in the prairies have been questioned by groups such as the Saskatchewan-based National Farmers Union (NFU 1983). They describe what they refer to as "chemical obsession" in the current agricultural system of the prairies. Some of them use relatively few or no pesticides and, through improved cultural techniques and better soil management, still farm competitively. We can only hope that the trend for a more rational use of insecticides will continue. One of the mandates of the Saskatoon Research Station of Agriculture Canada is to achieve a reduction in pesticide use while maintaining adequate crop protection (A. Ewen, pers. comm.). An even more fundamental question is whether crop protection is always achieved on a cost-effective basis.

4.3 Herbicide Use

The use of herbicides differs in a number of basic ways from the use of insecticides documented above. Herbicides are used much more generally and their use does not fluctuate greatly between years. The number of applications is usually fixed and predictable. Data on quantities of use and locations are more easily obtainable since the documentation is more complete and the pest populations more static.

Appendix 4.3 summarizes the use patterns of the main herbicides employed on prairie crops. For a more complete list as well as for information on permissible tank mixes, the reader is referred to the provincial spray schedules, published annually. Some differences do exist between provincial recommendations. For example, in contrast to Alberta, Manitoba and Saskatchewan do not recommend the use of 2,4-D on oats and flax. Manitoba does not recommend the use of barban on oil-seeds.

4.3.1 Types and quantities of herbicide use

Peden _et al_. (1982) presented an analysis of herbicide use obtained primarily from surveys of farmers by D.A. Dew (Alberta, 1976-77) and by A.G. Thomas (Manitoba, 1978-79 and Saskatchewan, 1977-79). Dew's data were based on a 20% return of the farmers surveyed and associated with 975 fields and Thomas' on a 50% return on 500 fields in each of the two provinces (Peden _et al_., op. cit.). Our analysis is largely based on the summary of these data by Peden _et al_. but is supplemented with data from Todd (1984) for an analysis of wild oat herbicides and is updated wherever possible with new crop estimates. We also drew on the data of Daciw (annual) and the Statistics Canada 1981 Census of Agriculture for comparison.

Herbicide use may be divided into two main categories: the control of broadleaf weeds such as smartweed and wild mustard and the control of grassy weeds such as green foxtail and wild oats. The proportion of major cereal and oil-seed crops treated for these two major weed categories has been estimated on the basis of the Thomas surveys conducted in Manitoba and Saskatchewan between 1977 and 1979 (Table 4.11).

Although herbicide use does not fluctuate as widely as insecticide use between years, the extent of use has not been static. Todd (1984) presented data from the Manitoba crop insurance surveys showing a dramatic increase in the use of wild oat herbicides between 1972 and 1982. Given Todd's 1982 estimates for Manitoba (Table 4.12), the estimates in Table 4.11 for grassy weed control appear to be underestimates. Todd estimated that 9.6 million ha of crop were treated for wild oats alone in 1982. Given the extensive nature of these data (although restricted geographically to Manitoba), we used them in preference to those contained in Table 4.11 in order to work out annual spray programs for grassy weeds. This may yield a slight overestimate of spray since the proportion of cropped hectares treated with wild oat herbicides was higher in Manitoba than in the other provinces. We assume, however, that wild oat control alone accounts for all of the grassy weed control activities in the prairies which counterbalances the overestimate.

From the proportion of crops sprayed yearly and the latest census data (1981, Statistics Canada), we calculated the total area sprayed with either broadleaf or grassy herbicides as 16.4 million ha (Table 4.13). Of course, there is overlap between the two types of herbicide spray because some undetermined proportion of the crops is treated for both types of weeds. According to Statistics Canada, the total area sprayed for weed and brush in the three prairie provinces in 1980 was 12.6 million ha. Another check is available through the 1982 Manitoba survey of prairie herbicide use (Daciw). Since quantities of actives only are supplied, a range of spray area was obtained depending on whether the highest or lowest label rates were used. This calculation was further complicated by several of the actives being combined for presentation. However, from our best calculations from these data, total prairie use of herbicides in 1981 would have ranged between 11.4 to 29.6 million ha with 15.4 million ha being the estimate corresponding to average application rates. Since this estimate is not critical for our assessment, the calculations will not be shown but Daciw's data are provided in Appendix 4.4. It is reassuring that the three estimates are quite close in estimating the total area sprayed annually. In comparison, the 1970 figure from Statistics Canada was 7.15 million ha, indicating an approximate increase of 76% between 1970 and 1980 given that comparable methodology was employed for both estimates.

4.3.2 Market share data

Data are also available on the identity and relative market share of the herbicides involved in the spray. Peden et al. (1982) summarized the information gathered by Dew and Thomas for seven of the most common broadleaf and grassy weed herbicides used on cereal crops (Table 4.14).

Table 4.11. Percent of crop areas treated for two major weed categories in Manitoba and Saskatchewan between 1977 and 1979[a].

	Wheat	Oats	Barley	Flax	Rape
Saskatchewan 1971					
Broadleaf Weeds	81	45	69	48	4
Grassy weeds	26	--	17	49	70
Manitoba 1978					
Broadleaf weeds	83	79	70	77	7
Grassy weeds	29	0	32	4	21
Manitoba 1979					
Broadleaf weeds	76	70	74	62	3
Grassy weeds	44	7	42	25	12
Averages					
Broadleaf weeds	80	65	71	62	4.7
Grassy weeds	33	4	30	26	34

[a] Source: A.G. Thomas in Peden et al. 1982.

Table 4.12. Proportion of cropped hectares in Manitoba treated with wild oat herbicides, 1982[a].

Crop	% treated with wild oat herbicides
Wheat	57.3
Barley	52.0
Flax	75.9
Rape-seed	91.5
Lentils	76.7
Peas	89.2
Sunflowers	81.2

[a] Source: Todd 1984.

Table 4.13. Estimated crop hectarage sprayed annually with broadleaf or grassy weed herbicides in the grassland-parkland area of the prairies as defined in Chapter 3.

Crop	1981 Area[a] (thousands of ha)	Proportion of Area Sprayed		Area Sprayed (thousands of ha)		Area Sprayed by Either Broadleaf or Grassy
		broadleaf[b]	grassy[c]	broadleaf	grassy	
Wheat	11,619	80%	57.3%	9,296	6,658	10,609
Oats	1,022	65%	--[e]	664	—	664
Barley	3,956	71%	52.0%	2,808	2,057	3,404
Rye	390	20%[f]	--	78	--	78
Flax	449	62%	75.9%	278	341	408
Canola	1,141	4.7%	91.5%	54	1,044	1,049
Tame Hay[g]	1,972	12%	--	227	--	227
Total	20,549	--	--	13,405	10,295	16,439

Sources:
[a] Table 3.24
[b] Table 4.11
[c] Table 4.12
[d] Calculated presuming independence of the proportions sprayed for broadleaf and grassy
[e] "-" indicates negligible or 0
[f] B.G. Todd (pers. comm.); data from 1984 crop insurance statistics for fall rye, the dominant prairie variety
[g] In the absence of any hard data, we used the "ball park" estimate provided by B.G. Todd (pers. comm.). He estimates that 70% of forage crops used for seed and 5% of forage crops used for forage are treated. Approximately 10% of the total forage crop is used for seed production.

Table 4.14. Percent of cereal crop area receiving different herbicides in different years[a].

	Alta. 1977	Sask. 1977	Sask. 1978	Sask. 1979	Man. 1978	Man. 1979
2,4-D	30.6	49.0	47.9	41.3	28.2	21.3
MCPA	11.6	4.6	4.3	3.4	16.4	12.1
Dicamba and mixes	15.7	13.7	17.2	19.9	8.5	9.8
Bromoxynil and mixes	11.2	4.6	6.0	9.7	7.9	12.3
Triallate	19.8	17.4	17.2	22.0	15.4	14.0
Trifluralin	3.8	1.8	5.5	0.1	17.2	1.8
Barban	6.5	2.8	1.3	1.9	2.9	3.1

[a] Source: D.A. Dew and A.G. Thomas in Peden _et al_. 1982.

Todd (1984) presented similar information from Manitoba on the herbicides most commonly used against wild oats (Table 4.15). Although we now can check these estimates using the EPS registrant survey, the confidentiality of this survey once again prevents us from disclosing the results. The only non-confidential check is the data of Daciw (1982) (Appendix 4.4) in which the only comparable information is the relative importance of MCPA to 2,4-D for broadleaf control. On the basis of total actives sold and taking into account the various application rates for formulations of the two actives, it would seem that the use of MCPA as presented in Table 4.14 was underestimated in 1977-78 and it further appears that the use of MCPA has since grown. The trend to increased MCPA use is not recent: between 1966 and 1972 the Alberta Environment Conservation Authority (1974b) reported an increase in the use of MCPA for that province from .615 to 1.02 million ha while 2,4-D use remained constant at approximately 1.96 million ha. We estimated from Daciw's data that the acreage treated with MCPA relative to that treated with 2,4-D in 1981 followed a 0.66:1 ratio. The growing popularity of MCPA is largely due to the use of MCPA/bromoxynil mixtures which are less phytotoxic to the crops than 2,4-D. Bromoxynil is being increasingly applied against 2,4-D- and MCPA-tolerant weeds. However, the herbicide 2,4-D in amine or low-volatile ester form is still the most prevalent herbicide for broadleaf control. Triallate (Avadex), followed closely by diclofop-methyl (Hoe-Grass), is currently the compound of choice for wild oat control. Trifluralin is the herbicide of choice in canola since it controls wild oats as well as broadleaf weeds in that crop. On the small southern Saskatchewan watershed surveyed by Waite et al. (1984), greater quantities of Triallate were used than all other herbicides combined. However, the total area sprayed was greater for broadleafed weeds.

4.3.3 Temporal overlap between herbicide use and duck breeding

In the case of insecticides, it was important to show the chronological relationship between spray and duck breeding since insecticide sprays are less likely than herbicides to have an impact beyond the spray season and their use is more erratic both spatially and temporally. Herbicides, on the other hand, are applied with regularity in a repetitive fashion year after year. Their impact on the weed species is similar whether they are applied pre- or post-emergence or as slow release formulations in the fall. Since the duck nesting season spans the crop growing season, we can expect any cover reduction resulting from herbicide use to have an impact on nesting (Chapter 7). In the case of shrubby species, which usually take longer to establish than grassy ones, herbicide impacts are expected to be long-term. For those herbicides which are more likely to have an impact on an aquatic resource such as the invertebrate populations (see Chapter 7), the timing of application will be of greater concern. Spring-applied liquid herbicides still represent the bulk of the herbicide market, although trends indicate that fall-applied granulars are increasing in popularity. The latter help spread the farmer's workload more evenly and allow for earlier planting since they eliminate the need to till for wild oat control (Zittlau 1979). As such, these herbicides may be a positive factor in reducing nest losses to tillage (Chapter 2).

Table 4.15. Market share of wild oat herbicides in Manitoba in 1982[a].

Crop	% of Area Treated with a Given Herbicide									
	Triallate	Diclofop-Methyl	Difenzoquat	Flamprop-Methyl	Barban	Trifluralin	Asulam	EPTC	Sethoxydim	Triallate and Trifluralin
Wheat	29.1%	21.3%	3.9%	2.2%	0.6%	-[b]	-	-	-	-
Barley	29.3%	15.1%	7.0%	-	0.6%	-	-	-	-	-
Flax	9.8%	15.6%	-	-	0.2%	21.3%	19%	8.2%	1.1%	-
Rape-seed	1.4%	8.5%	-	-	0.6%	73.3%	-	-	0.2%	7.4%

[a] Source: Todd 1984

[b] "-" indicates negligible or 0.

4.3.4 Factors affecting herbicide use

4.3.4.1 Pest distribution

Weeds have a much more static distribution than insect pests and hence easily lend themselves to a mapping exercise. Based on weed density maps of Thomas and Dew, Peden et al. (1982) have mapped the combined areas of highest density for the main broadleaf and grassy weeds by province (Appendix 4.5). The only generality which emerges from the maps is the close association of wild oats with the Parklands, which had already been noted by Todd (1984) for Manitoba.

4.3.5 Ultimate factors, cost-benefit of herbicide use

Possibly the single most important factor which will dictate future uses of herbicides is the implementation of no-till or at least low-till strategies. The implications of reduced tillage cropping practices on herbicide use have recently been reviewed by A.A. Aquatic Research Ltd. (1985) under contract with EPS, Western and Northern Region. The following section is largely based on this review. In the U.S., reduced tillage cropping has been gaining in popularity (a doubling in area between 1973 and 1981), especially in areas suffering from high soil erosion problems, such as the midwest. No-till cropping, on the other hand, has not yet become a popular strategy. There has been a similar trend to reduced tillage in Canada, much of which can be ascribed to a reduction in summer fallow and an increased reliance on herbicides. Experts predict that this trend will continue but reduced-tillage may only be viable in parts of the prairies (A.A. Aquatic Research Ltd. 1985). Factors which mitigate against reduced-tillage are the high costs of herbicides, the need for specialized seeding equipment and poor farm management. It is currently estimated that only 50,000 ha are under a no-till regime in the prairies. Although it is generally recognized that reduced-tillage will result in a heavier use of herbicides, the extent of the increase is debatable so that no estimate can be given.

A shift in the type of herbicide employed is also expected since reduced-tillage cropping tends to favor perennial weeds such as quack grass and Canada thistle (A.A. Acquatic Research Ltd. 1985). Reduced tillage cropping is likely to result in an increased use of broad-scale, fast-acting but low residual herbicides such as paraquat and glyphosate. An increased reliance on post-emergence herbicides is also predicted. New herbicides such as fluazifop butyl (Fusilade) and sethoxydim (Poast) for the control of most annual and perennial grasses and clopyralid (Lontrel, presently in the registration process) for annual and perennial broadleaf weed control in canola crops are seen as breakthroughs for reduced tillage cropping.

Chemical fallow is a subset of reduced tillage farming which may also increase in popularity in view of anticipated developments in chemical fallow herbicides (Rennie et al. 1984). The primary impediment to a more widespread acceptance by farmers is currently the cost of the herbicides.

However, although chemical fallow has been shown to provide excellent control of wind and water erosion and has been shown to result in a greater capture of rain and snow, the high losses of N associated with this practice are of serious concern (Ibid). In the long-term, given the steady degradation of our western soils (Coote 1983, Standing Senatorial Committee on Agriculture, Fisheries and Forestry 1984), there is little doubt that current practices of land management in western Canada will have to change.

4.4 Other Chemical Uses

Other pesticides used extensively in cereal and oil-seed crops are fungicides. Up to now, they have been used almost entirely as seed coatings to control seed-borne diseases such as smut and to prevent seedling blights and seed decay (Hay 1980). In areas of the world with higher humidity (e.g. Great Britain), post-emergence fungicides have become an integral part of cereal culture. Several post-emergence fungicides for use on cereals are currently going through the Canadian registration system and, therefore, growth in this area now seems inevitable. Currently, most of the use of seed treatment is for oil-seeds. Hay (1980) estimated that 80% of rape-seed, 50% of flax and rye and 10% of wheat, barley and oat seeds are currently treated with fungicides. The fungicides currently registered for use as seed treatments are given in Table 4.16. Seed treatments are usually applied as mixtures of several fungicides and lindane (see below). Their use on cereals is markedly lower than what it was 20 years ago as a result of incorporated resistance in new cultivars.

The possible impacts of fungicides on prairie ecosystems have yet to be demonstrated. Given the importance of the detritivore pathways in wetland nutrient dynamics and the usual role of fungi in this respect (Chapter 2), we may expect disturbances at the whole ecosystem level. Furthermore, some fungicides are known to have herbicidal activity so that the assessment of their possible impacts on prairie ecosystems might parallel the assessments of herbicide impacts (Chapter 7). One fungicide, whose identity must, for the time being, remain confidential since it is presently undergoing registration, has been shown to be a more efficient algicide than the few herbicides already registered for that purpose.

The estimated 4,500 kg of lindane used annually as a seed treatment to reduce damage by wireworms (Hay 1980) was not discussed in the insecticide section. An unknown quantity is also used against flea beetles in rape. Lindane became available in 1973 and replaced other organochlorines such as aldrin, dieldrin and heptachlor (Alberta Environment Conservation Authority 1974a) which had been shown to lead to widespread contamination of the environment. Given a poor correlation between the use of insecticide seed protectants and cereal yields in the mid-60's, the same source (AECA 1974a) questioned the wisdom of using any insecticide on cereal seeds.

Finally, this section would not be complete without mention of the other widely used agricultural chemicals not covered under the Pest Control

Table 4.16. Fungicides currently registered as seed treatments in oilseed, cereal and forage crops[a].

Active	Oil-seeds			Cereals				Forage		
	Canola	Flax	Mustard	Wheat	Oats	Barley	Rye	Alfalfa	Clover	Grasses
Benomyl	X		X							
Captan	X		X					X	X	X
Formaldehyde				X	X					
Maneb		X		X	X	X	X			
Mancozeb		X		X	X	X				
Metiram		X		X	X	X				
2-(Thiocyanomethyl thio) Benzothiazole		X		X	X	X				
Thiram	X	X	X	X	X	X	X	X	X	
Carbathiin	X	X	X	X	X	X	X			

[a] Source: Agriculture Canada 1984.

Product Act - fertilizers. The use of fertilizers in western Canada (including British Columbia) was summarized by Rennie et al. (1980) (Table 4.17). Of note is the large increase over time in the demand for nitrogen and, to a lesser extent, phosphorus. It is estimated that, in order to curtail the rate of organic matter loss from prairie soils, nitrogen fertilizer consumption will increase by 300% in the 1980's (Ibid). According to the 1981 Census of Canada (Statistics Canada), 69%, 46% and 59% of croplands were fertilized in Manitoba, Saskatchewan and Alberta, respectively. The possible impact from eutrophication as a result of large quantities of nutrients finding their way into aquatic prairie ecosystems is outside the scope of this report but is an issue which deserves attention.

Table 4.17. Fertilizer use in western Canada for selected years between 1938 and 1979[a].

Year	Fertilizer Used (tonnes x 10^3) N	P_2O_5	K_2O	S[b]
1938	0.9	3.6	-[c]	-
1948	6.4	24.5	-	-
1958	14.5	29.9	-	-
1968	191.4	231.3	-	-
1969	127.0	146.1	5.4	18.8
1970	108.0	102.5	6.4	25.7
1971	152.4	147.0	8.2	35.2
1972	163.3	153.3	12.7	33.2
1973	239.5	217.7	8.2	32.0
1974	295.7	290.3	9.1	43.0
1975	329.3	310.3	15.4	44.0
1976	362.0	274.9	18.1	38.0
1977	368.3	253.1	26.3	36.6
1978	480.8	323.9	31.8	36.6
1979	591.5	374.7	41.7	39.6

[a] Source: Rennie *et al*. 1980

[b] Mainly incidental additions but the 1976-79 values include consumption of materials used specifically to supply sulphur

[c] "-" indicates negligible or 0.

Appendix 4.1. Recent (1985) information from the Manitoba Crop Insurance Corporation on market share data for herbicides used in Manitoba. These reports were received too late to be incorporated in the text but they are given here as an indication of what they have to offer. They represent the most comprehensive and detailed analysis of herbicide use available for any prairie province.

WEED REFERENCE BULLETIN

MANITOBA
AGRICULTURE
Weed Section

CROPS July 1985

HERBICIDE USE ON HARD RED SPRING WHEAT IN MANITOBA

Data from Manitoba Crop Insurance Corporation questionnaires has been summarized in Tables 1 and 2. This data reveals trends in herbicide use from 1981 to 1984 and illustrates differences in herbicide use in each of the five regions of the province.

1. 1981 to 1984 Trends

Use of Avadex either alone or in combination with Treflan declined dramatically during this period. During the same period the acreage treated with Avenge nearly tripled and Hoe-Grass treated acreage also increased significantly. During this time the percentage of the wheat acreage treated with wild oat herbicide increased from 56.8% to 61.3%.

Broadleaf weed control has shifted away from the use of 2,4-D or MCPA in favour of broader spectrum products, particularly bromoxynil + MCPA formulations. Use of DyVel, Estaprop, Target and Tordon 202 C increased steadily from 1981 to 1984.

2. Herbicide Use by Region

Greatest use of soil incorporated herbicides continues to be in the Southwest region.

Avenge use is focused in the Eastern half of the province while Hoe-Grass predominates the postemergence wild oat market on the western side.

Use of Estaprop is highest in the Southwest, probably in association with kochia control. DyVel is most popular in the Northwest region, being utilized for hemp nettle control.

Use of bromoxynil + MCPA products is greatest in the Eastern, Central and Interlake regions. In these areas both wild buckwheat and annual smartweed species preclude the use of phenoxy herbicides alone. In the Southwest, use of Banvel and/or 2,4-D is common while in the Northwest region farmers prefer MCPA.

Table 1. Herbicide Use in Red Spring Wheat in Manitoba 1981 to 1984

Herbicide	% of Total Acreage Treated			
	1981	1982	1983	1984
Avadex BW	10.37	7.84	6.53	5.60
Avadex BW + Treflan EC	21.48	21.29	17.71	13.17
Treflan EC	2.85	1.97	2.04	1.80
Afolan or Lorox	1.36	1.17	0.88	0.81
Avenge 200 C	3.69	3.86	6.13	10.65
Banvel	11.18	13.62	14.61	12.67
Blagal	0.41	0.35	0.20	0.34
Brominal M/Buctril M/Sabre	9.29	12.16	15.27	19.54
Carbyne 2 EC	1.21	0.63	0.64	0.33
DyVel	2.04	2.68	3.34	3.98
Estaprop	2.22	2.62	3.58	3.85
Glean	--	0.83	0.21	0.26
Hoe-Grass 284	16.64	21.34	21.27	20.19
Hoe-Grass II	--	--	3.09	8.72
Kil-Mor	0.08	0.04	0.05	0.03
Lexone/Sencor	0.03	0.05	0.05	0.13
Mataven	3.40	2.17	2.45	2.65
MCPA	25.24	24.72	20.90	20.10
MCPAK	1.09	0.92	0.97	0.81
Pardner/Torch/Torch DS	8.07	9.52	5.29	3.61
Stampede 360	1.34	0.56	0.71	0.76
Stampede CM	4.36	4.28	3.49	3.37
Target	0.56	0.73	0.92	1.23
Tordon 202 C	2.15	2.73	3.55	3.27
2,4-D	35.23	33.51	31.39	25.91
Untreated	7.00	4.41	5.25	5.20

Table 2. 1984 Herbicide Use in Red Spring Wheat by Region

Herbicide	Central	Eastern	Interlake	Southwest	Northwest	All Manitoba
Avadex BW	3.83	3.48	6.93	4.48	9.83	5.60
Avadex BW + Treflan EC	10.99	13.83	6.39	18.37	7.60	13.17
Treflan EC	1.12	0.52	0.28	3.46	0.16	1.80
Afolan or Lorox	0.47	0.17	0.11	0.55	2.10	0.81
Avenge 200 C	15.31	20.63	22.70	5.51	8.60	10.65
Banvel	9.80	6.22	7.40	16.15	12.14	12.67
Blagal	0.04	--	0.33	0.07	1.48	0.34
Brominal M/Buctril M/Sabre	33.45	37.12	23.91	9.01	13.56	19.54
Carbyne 2 EC	0.04	2.26	0.41	0.13	0.71	0.33
DyVel	2.25	4.86	2.22	1.77	11.86	3.98
Estaprop	1.68	--	5.58	6.84	1.70	3.85
Glean	0.12	--	0.16	0.05	1.06	0.26
Hoe-Grass 284	19.50	13.94	15.81	17.78	29.44	20.19
Hoe-Grass II	6.32	5.73	12.69	7.56	14.91	8.72
Kil-Mor	0.06	0.31	--	--	--	0.03
Lexone/Sencor	0.08	--	0.07	0.02	0.49	0.13
Mataven	2.79	8.96	4.37	1.83	2.06	2.65
MCPA	19.19	22.09	23.55	16.11	27.75	20.10
MCPAK	0.06	0.61	1.70	0.48	2.58	0.81
Pardner/Torch/Torch DS	3.59	3.03	5.00	3.44	3.82	3.61
Stampede 360	0.98	0.01	0.62	0.88	0.40	0.76
Stampede CM	4.72	0.93	1.22	3.84	1.44	3.37
Target	0.52	0.34	0.87	1.60	1.69	1.23
Tordon 202 C	1.27	0.33	2.80	3.03	7.77	3.27
2,4-D	17.47	7.95	17.79	42.90	10.82	25.91
Untreated	8.71	14.35	5.98	2.81	1.50	5.20

WEED REFERENCE BULLETIN

MANITOBA
AGRICULTURE
Weed Section

CROPS August 1985

HERBICIDE USE ON FLAX IN MANITOBA

Data from Manitoba Crop Insurance Corporation questionnaires has been summarized in Tables 1 and 2. This data reveals trends in herbicide use from 1981 to 1984 and illustrates the differing patterns of herbicide use in each of the five regions of the province.

1. 1981 to 1984 Trends

This data illustrates the dramatic rise in the use of Poast following its introduction in 1982. Over half of all flax acres in the province are now treated with this one herbicide. Use of all other herbicides for grassy weed control has declined as a result. Approximately 82% of the flax acreage was treated for wild oat control in 1984.

Broadleaf weed control has continued to shift towards use of products containing bromoxynil + MCPA (Brominal M/Bromox 450 M/Buctril M/Sabre). Over half the flax acreage is now treated with these herbicides. Consequently, use of MCPA alone has declined. These trends undoubtedly reflect the continuing increase in annual smartweed infestations in flax.

2. Herbicide Use by Region

Use of Treflan on flax continues to be greatest in the Southwest region. In all other regions, use of soil incorporated herbicides is significantly lower, while use of postemergence herbicides is correspondingly greater. In the Southwest, straight MCPA is still a popular treatment (fewer smartweed problems) while in all other regions farmers rely heavily on bromoxynil + MCPA combinations for broadleaf weed control.

Table 1. Herbicide Use in Flax in Manitoba 1981 to 1984.

Herbicide	% of Flax Acreage Treated			
	1984	1983	1982	1981
Asulox F	3.42	8.37	19.01	16.16
Avadex BW	1.25	2.47	3.55	3.60
Avadex BW + Treflan	2.23	4.81	6.23	8.30
Basagran	0.02	0.06	0.02	0.04
Basfapon/Dalapon/Dowpon	0.25	0.93	1.26	1.51
Blagal	0.46	0.28	0.30	0.58
Brominal M/Bromox 450 M/ Buctril M/Sabre	52.77	49.85	41.78	33.14
Eptam	1.71	3.94	8.18	6.71
Fusilade	1.28	-	-	-
Hoe-Grass 284	4.65	9.51	15.82	11.68
Hoe-Grass II	1.58	1.22	-	-
MCPA (amine and ester)	19.05	21.57	26.21	28.93
MCPA sodium salt	1.58	2.47	2.55	2.97
MCPAK	0.27	0.23	0.40	0.17
Poast	52.92	30.93	1.08	0.18
Stampede 360	0.21	0.15	0.18	0.41
Stampede CM	0.34	0.63	1.22	0.92
TCA	1.14	1.54	1.68	2.41
Torch/Torch DS	1.36	2.99	4.82	4.21
Treflan (1/2 rate)	1.67	3.09	2.20	3.16
Treflan (full rate)	12.98	13.90	21.39	24.65
Untreated	6.96	7.57	6.67	11.18

Table 2. 1984 Herbicide Use in Flax by Region

Herbicide	% of Flax Acreage Treated					
	Central	Eastern	Interlake	Southwest	Northwest	All Manitoba
Asulox F	4.09	6.51	7.05	0.99	5.73	3.42
Avadex BW	0.76	1.41	3.58	1.52	1.92	1.25
Avadex BW + Treflan	2.08	1.77	0.85	2.51	3.41	2.23
Basagran	0.03	-	-	-	-	0.02
Basfapon/Dalapon/Dowpon	0.20	-	-	0.41	0.13	0.25
Blagal	0.31	-	0.40	0.31	2.82	0.46
Brominal M/Bromox 450 M/ Buctril M/Sabre	63.65	51.78	51.06	34.03	56.07	52.77
Eptam	0.62	-	0.47	2.69	7.86	1.71
Fusilade	0.62	2.53	2.70	1.21	5.32	1.28
Hoe-Grass 284	3.76	2.41	4.07	6.52	5.32	4.65
Hoe-Grass II	0.72	2.09	5.35	2.30	2.69	1.58
MCPA (amine and ester)	11.96	11.08	22.06	33.29	12.86	19.05
MCPA (sodium salt)	1.76	2.77	1.14	1.31	0.91	1.58
MCPAK	0.06	-	0.88	0.62	0.30	0.27
Poast	62.01	58.43	45.96	38.39	47.94	52.92
Stampede 360	0.22	-	0.59	0.20	0.24	0.21
Stampede CM	0.31	0.27	-	0.51	-	0.34
TCA	0.53	-	0.94	2.58	0.29	1.14
Torch/Torch DS	0.90	1.08	6.39	1.30	2.82	1.36
Treflan (1/2 rate)	0.47	0.42	0.82	4.30	0.25	1.67
Treflan (full rate)	9.70	1.48	0.35	23.83	4.33	12.98
Untreated	8.37	13.34	6.02	3.77	3.96	6.96

Appendix 4.2. Insecticides recommended in 1984 for selected crops and pests in the three prairie provinces[a]. An "X" refers to a recommendation for all crop species within a crop type unless otherwise indicated. An "*" indicates that the product is not registered but is otherwise acceptable to the WCCP[b].

Crop	Pest	Insecticide	Rate (g.ai/ha)	Alberta	Saskatchewan	Manitoba	Economic Threshold Provided
Cereals Barley (B), Oats (O), Rye (R), Wheat (W)	aphids	Demeton	210	X (B,O,W)	X		yes
		Dimethoate	210	X	X	X (B)	
		Malathion	700	X	X	X	
	armyworms	Carbaryl	1125-2250	X	X[c]	X	yes
		Malathion	560-1400	X	X	X	
		Methomyl	340	X (B,O,W)	X	X (B,O,W)	
		Trichlorfon	560	X (B,O,W)	X	X (B,O,W)	
		Deltamethrin	7.5-10	*			
	thrips	Dimethoate	470-500	X			no
		Malathion	560	X			
	cutworms (army, pale western, red-backed)	Chlorpyrifos	420-560	X	X	X	yes
		Permethrin	70-150		X	X	
		Deltamethrin	7.5-10	*			
	grasshoppers	Carbaryl	550-1125	X	X	X	yes
		Carbofuran	140	X (B,O,W)	X (B,O,W)	X	
		Deltamethrin	5-7.5		X	X	
		Dimethoate	210-490	X	X		
		Malathion	840-850	X	X		
	wireworms	Lindane (seed treat.)	0.5 g ai/kg seed	X	X	X	not definite
	say stink bug	Dimethoate	275	X			yes

Appendix 4.2 (cont'd)

Crop	Pest	Insecticide	Rate (g.ai/ha)	Alberta	Saskatchewan	Manitoba	Economic Threshold Provided
Oil-seeds Canola (C), Flax (F), Mustard (M), Sunflowers (S)	alfalfa looper	Chlorpyrifos	350-490	X (C)			no
		Methomyl	200-250	X (C)			
	aphids	Dimethoate	280-350	X (C)	X (C,F)		no
	cutworms (army, pale western, red-backed)	Clorpyrifos	420-560	X	X	X	yes
		Permethrin	70-150			X (C,M,S)	
		Deltamethrin	7.5-10	* (F)			
	beet webworm	Chlorpyrifos	360			X (C,M)	no
		Methomyl	250	X (C,F)	X (C,F)	X (C,F)	
		Methamidophos	264-528			X (C,M)	
		Trichlorfon	420	X (C,F)	X (C,F)		
	bertha armyworm	Chlorpyrifos	350-490	X (C)	X (C)	X (C,M)	yes
		Methamidophos	264-560	X (C)	X (C)	X (C,M)	
		Methomyl	250	X (C,F)	X (C,F)	X (C,F,M)	
		Deltamethrin	5-7.5	* (C,M)			
	clover cutworms	Chlorpyrifos	360			X (C,M)	yes
		Methamidophos	264-528			X (C,M)	
		Methomyl	250	X (C)	X (C)	X (C,M)	
		Deltamethrin	5-7.5	* (C,F)			
	diamond-back moth	Azinphos-methyl	350	X (C)	X (C)	X (C,M)	yes
		Malathion	350	X (C,M)	X (C,M)	X (C,M)	
		Methidathion	260	X (C)	X (C,M)		
		Trichlorfon	1120-1176	X (C)	X (C)	X (C,M)	
	flax bollworm	Methomyl	250	X (F)	X (F)		yes

Appendix 4.2 (cont'd)

Crop	Pest	Insecticide	Rate (g.ai/ha)	Alberta	Saskatchewan	Manitoba	Economic Threshold Provided
Oil-seeds Canola (C), Flax (F), Mustard (M), Sunflowers (S)	flea beetle (seed treat.)	Carbofuran (granular)	225-280	X (C,M)	X (C,M)	X (C,M)	no
		Lindane	16 g/kg seed	X (C,M)	X (C,M)	X (C,M)	
		Terbufos (granular)	280-560	X (C,M)	X (C,M)	X (C,M)	
		Isofenphos	.3 g/kg seed	* (C)			
		Endosulfan	560	* (C)			
	flea beetles (foliar sprays)	Azinphos-methyl	70	X (C)	X (C)	X (C,M)	yes
		Carbaryl	500-1125	X (C)	X (C,M)	X (C,M)	
		Carbofuran	70-140	X (C,M)	X (C,M)	X (C,M)	
		Deltamethrin	5-7.5	X (C,M)	X (C,M)	X (C,M)	
		Malathion	560	X (C,M)	X (C,M)	X (C,M)	
		Methidathion	160	X (C,M)	X (C,M)		
	grasshoppers	Carbaryl	560	X (C)			yes
		Carbofuran	140	X	X	X (C,M)	
		Malathion	840	X (C,F,M)	X (C,F,M)	X (C,M)	
		Methamidophos	560	X (C,F,M)	X (C)		
		Cypermethrin	20-28	* (C)			
	painted lady butterfly	Methidathion	525-700	X			
	red turnip beetle	Azinphos-methyl	140-200	X (C)	X (C)	X (C,M)	no
		Carbofuran	140	X (C,M)	X (C,M)	X (C,M)	
		Methidathion	260	X (C,M)	X (C,M)		
	sunflower beetle	Azinphos-methyl	280	X (S)	X (S)	X (S)	yes
		Carbofuran	70-140	X (S)	X (S)	X (S)	
		Endosulfan	560	X (S)	X (S)	X (S)	
		Methidathion	260-525	X (S)	X (S)		
	thistle caterpillar	Azinphos-methyl	280			X (S)	no

Appendix 4.2 (cont'd)

Crop	Pest	Insecticide	Rate (g.ai/ha)	Alberta	Saskatchewan	Manitoba	Economic Threshold Provided
Forage crops Alfalfa (A), Clover (C), Grasses (G)	alfalfa weevil	Azinphos-methyl	560-840	X (A)			
		Carbofuran	260-375	X (A)			
		Malathion	1000-1400	X (A)			yes
		Methidathion	280-560	X (A)			
		Methoxychlor	1125-1400	X (A)			
		Phosmet	1125	X (A)			
	pea aphid	Demeton	280-325	X (A)	X (A)		
		Dimethoate	210	X (A)	X (A)		
		Malathion	700	X (A)	X (A)		yes
		Methidathion	280-560	X (A)	X (A)		
		Deltamethrin	7.5-10	* (A)			
	grasshoppers	Azinphos-methyl	140-420	X (A,C)	X (A,C)		
		Carbaryl	550-1125	X		X	
		Carbofuran	140	X	X	X	yes
		Dimethoate	210-420	X	X		
		Malathion	840	X	X	X	
		Deltamethrin	5-6	* (G)			
	plant bugs (alfalfa plant bug, superb plant bug, lygus bug, Plagiognathus bug)	Dimethoate	560	X (A)	X (A)		
		Methidathion	280-560	X (A)	X (A)	X (A)	yes
		Trichlorfon	560	X (A)	X (A)	X (A)	
		Deltamethrin	7.5-10	* (A)			
	sweet clover weevil	Azinphos-methyl	420	X (C)	X (C)	X (C)	
		Carbaryl	1125	X (C)	X (C)	X (C)	yes
		Dimethoate	560	X (C)	X (C)		
		Malathion	700	X (C)			

Appendix 4.2 (cont'd)

[a] Sources: WCCP 1984, Saskatchewan Agriculture 1984, Manitoba Agriculture 1984

[b] WCCP (Western Committee on Crop Pests) consists of provincial prairie agriculture departments as well as federal agriculture officials and trade organizations. Makes recommendations which may or may not be accepted by all provinces in their own publications

[c] There appears to be an error in the Saskatchewan recommendations which reads "125-250 g.ai/ha".

Appendix 4.3. Herbicides recommended by one or more provinces between 1982 and 1985, primarily for the control of A) broadleaf weeds B) grassy weeds C) both broadleaf and grassy weeds or as a dessicant[a].

A)	Cereals				Oil-seeds				Forage		
	Wheat	Oats	Rye	Barley	Canola	Mustard	Flax	Sunflowers	Alfalfa	Clover	Grasses
amitrole									X	X	
benazolin					X						
bentazon							X				
bromoxynil	X	X	X	X			X			X	
chloramben								X			
clopyralid[b]					X						
cyanazine	X	X		X			X				
2,4-D	X	X	X	X			X				X
2,4-DB	X	X		X	X				X	X	X
dicamba	X	X		X							X
dichlorpop	X			X							
glyphosate									X	X	X
linuron	X	X		X			X				
MCPA	X	X	X	X			X		X	X	X
MCPB	X	X	X	X						X	X

Appendix 4.3 (cont'd)

A) (cont'd)	Cereals				Oil-seeds				Forage		
	Wheat	Oats	Rye	Barley	Canola	Mustard	Flax	Sunflowers	Alfalfa	Clover	Grasses
mecoprop	X	X		X							
metribuzin	X			X							
picloram	X			X							
propanil	X	X		X			X				

B)	Cereals				Oil-seeds				Forage		
	Wheat	Oats	Rye	Barley	Canola	Mustard	Flax	Sunflowers	Alfalfa	Clover	Grasses
asulam							X		X		
barban	X			X	X	X	X	X	X	X	X
dalapon					X		X		X		
diclofop-methyl	X		X	X	X	X	X	X	X	X	X
difenzoquat	X		X	X					X	X	X
flamprop-methyl	X							X	X		X
fluazifop-butyl[b]							X	X	X	X	

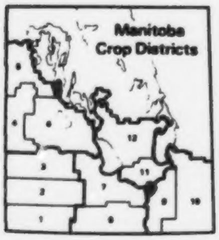
Appendix 4.3 (cont'd)

B) (cont'd)	Cereals				Oil-seeds				Forage		
	Wheat	Oats	Rye	Barley	Canola	Mustard	Flax	Sunflowers	Alfalfa	Clover	Grasses
propyzamide									X		X
sethoxydim					X		X				
sodium TCA		X		X	X		X				
triallate	X			X	X	X	X		X	X	

C)	Cereals				Oil-seeds				Forage		
	Wheat	Oats	Rye	Barley	Canola	Mustard	Flax	Sunflowers	Alfalfa	Clover	Grasses
diquat					X	X	X	X	X	X	
EPTC							X	X	X		
simazine									X		
terbacil									X		
trifluralin	X			X	X	X	X	X			

[a] Sources: Saskatchewan Agriculture 1983, Alberta Agriculture 1982, Manitoba Agriculture 1985
[b] Recent and/or temporary registrations.

Appendix 4.4. Example of the information available through the annual registrant survey published by Manitoba on an annual basis (from Daciw 1982).



AGRICULTURE STATISTICS

☐Crops ☐Livestock ☐Finance ☐Farms

Economics Branch, 903 Norquay Building,
Winnipeg, Manitoba R3C 0V8

Subject: HERBICIDES USED AGRICULTURALLY IN WESTERN CANADA FOR WEED CONTROL, 1980-81

TOTAL WEIGHT USED ACCORDING TO FORMULATIONS

	2,4-D							
	ESTER		ESTER (Low Vol.)		AMINE		TCA	
	1980	1981	1980	1981	1980	1981	1980	1981
Prov.	000 lbs							
Man.	38.6	25.2	111.6	128.3	839.7	866.5	600.9	358.9
Sask.	1,478.5	2,346.5	1,043.4	1,253.8	1,070.1	1,510.4	481.8	328.8
Alta.	804.5	692.8	443.5	378.6	576.4	675.9	285.5	151.2
B.C.	1.2	1.9	4.4	1.0	16.1	13.4	20.0	-
	2,322.8	3,066.4	1,602.9	1,761.7	2,502.3	3,066.2	1,388.2	838.9

	MCPA							
	ESTER		AMINE		Na + K SALT		DALAPON	
	1980	1981	1980	1981	1980	1981	1980	1981
	000 lbs							
Man.	98.7	152.3	888.7	822.7	126.9	144.5	47.4	26.1
Sask.	220.6	351.6	726.3	982.3	60.2	86.2	9.1	56.3
Alta.	565.3	558.4	1,194.7	1,131.9	238.9	223.3	0.5	0.6
B.C.	3.0	5.2	14.6	10.5	-	1.2	0.5	2.2
	887.6	1,067.5	2,824.3	2,947.4	426.0	455.2	57.5	85.2

	OTHER CHEMICALS*		WILD OAT HERBICIDES**	
	1980	1981	1980	1981
	000 lbs		acres treated	
Man.	544.3	1,111.7	4,437,600	4,586,600
Sask.	752.7	1,705.1	8,100,100	6,999,300
Alta.	629.0	1,181.6	5,059,100	6,038,900
B.C.	41.5	42.6	146,800	154,600
	1,967.5	4,041.0	17,743,600	17,779,400

---

* Bromoxynil, Dicamba, Dichloroprop, Linuron, Roundup, Stampede, Tordon 202C and others.

** Asulox, Atrazine, Avadex, Avenge, Carbyne/Wypout, Eptam, Eradicane, Hoe-grass, Mataven and Treflan.

The cooperation received from all the chemical firms and distributors who supplied the data on the amounts of herbicides used in each province is greatly appreciated.

M. Daciw, Sr. Statistical Analyst
Ph: (204) 944-3496

January 7, 1982

MANITOBA
AGRICULTURE

TOTAL WEIGHT USED ACCORDING TO FORMULATIONS

	2,4-D							
	ESTER		ESTER (Low Vol.)		AMINE		TCA	
	1980	1981	1980	1981	1980	1981	1980	1981
Prov.	-000 kg-							
Man.	17.5	11.4	50.6	58.2	380.9	393.0	272.6	162.8
Sask.	670.6	1 064.4	473.3	568.7	485.4	685.1	218.5	149.1
Alta.	364.9	314.2	201.2	171.7	261.4	306.6	129.5	68.6
B.C.	0.6	0.9	2.0	0.5	7.3	6.1	9.1	-
	1 053.6	1 390.9	727.1	799.1	1 135.0	1 390.8	629.7	380.5

	MCPA							
	ESTER		AMINE		Na + K SALT		TCA	
	1980	1981	1980	1981	1980	1981	1980	1981
	-000 kg-							
Man.	44.8	69.1	403.1	373.2	57.6	65.6	21.5	11.8
Sask.	100.1	159.5	329.5	445.5	27.3	39.1	4.1	25.5
Alta.	256.4	253.3	541.9	513.4	108.4	101.3	0.2	0.3
B.C.	1.4	2.3	6.6	4.8	-	0.5	0.2	1.0
	402.7	484.2	1 281.1	1 336.9	193.3	206.5	26.0	38.6

	OTHER CHEMICALS*		WILD OAT HERBICIDES**	
	1980	1981	1980	1981
	-000 kg-		-hectares treated-	
Man.	246.9	504.3	1 795 800	1 856 100
Sask.	341.4	773.4	3 278 000	2 832 500
Alta.	285.3	536.0	2 047 400	2 443 900
B.C.	18.8	19.3	59 400	62 600
	892.4	1 833.0	7 180 600	7 195 100

---

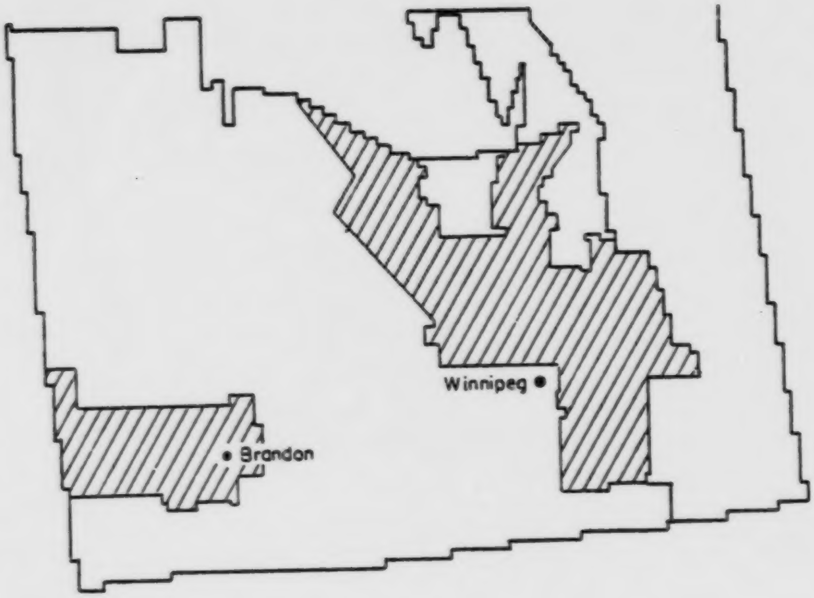
* Bromoxynil, Dicamba, Dichloroprop, Linuron, Roundup, Stampede, Tordon 202C and others.

** Asulox, Atrazine, Avadex, Avenge, Carbyne/Wypout, Eptam, Eradicane, Hoe-grass, Mataven and Treflan.

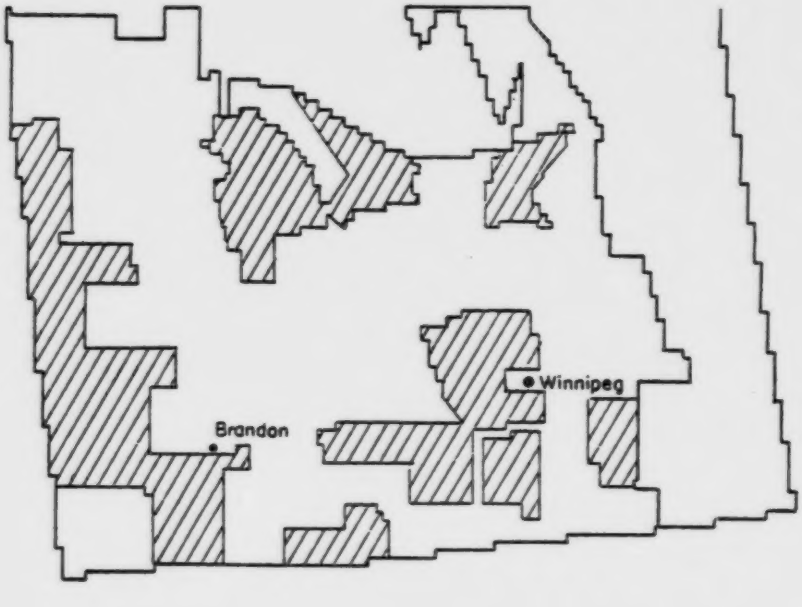
- -

Page 1 contains data expressed in the Imperial System of measurements. Page 2 contains the same data converted to the International System of Units. The conversion factors used were: one pound = 453.5923 grams and one acre = 0.40468564 hectares.

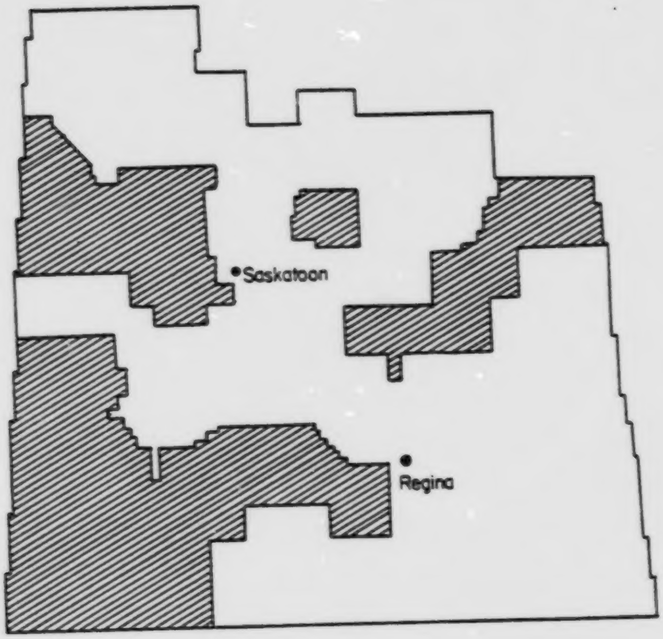
Appendix 4.5. Distribution maps for the major broadleaf and grassy weeds in all three prairie provinces (from Peden *et al*. 1982).


Winnipeg
Brandon

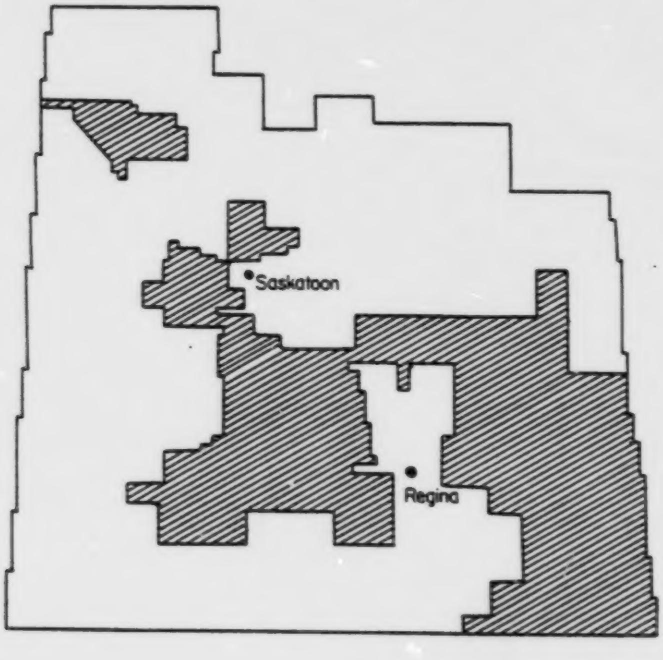
Areas of highest densities of Canada Thistle, Wild Mustard, and Smartweed in Manitoba, 1979.


Winnipeg
Brandon

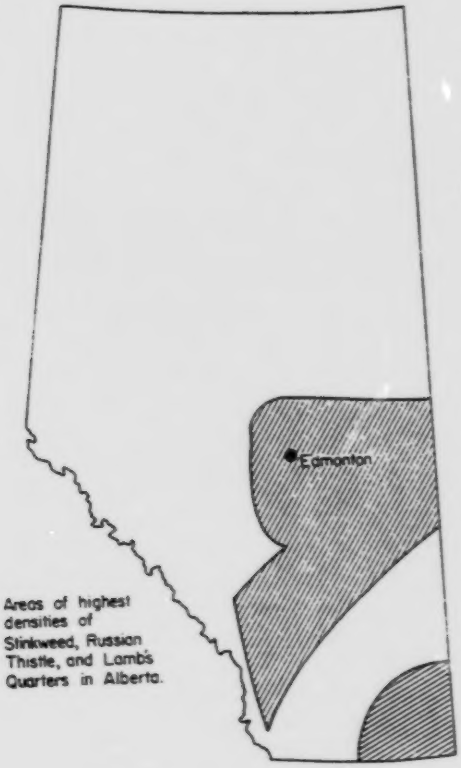
Areas of highest densities of Wild Oats, Green Foxtail, and Wild Buckwheat in Manitoba, 1979.


Saskatoon
Regina

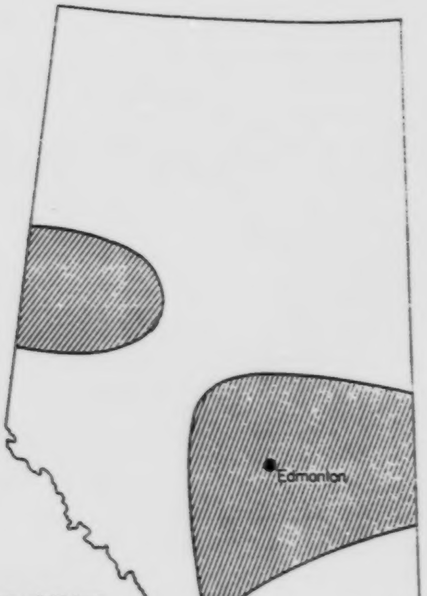
Areas of highest densities of Russian Thistle, Stinkweed, and Lamb's Quarters in Saskatchewan, 1979.


Saskatoon
Regina

Areas of highest densities of Wild Oats, Green Foxtail, and Wild Buckwheat in Saskatchewan, 1979.



Areas of highest densities of Stinkweed, Russian Thistle, and Lamb's Quarters in Alberta.



Areas of highest densities of Green Foxtail, Wild Oats, and Wild Buck Wheat in Alberta.

CHAPTER 5

LIKELIHOOD OF PESTICIDE CONTAMINATION OF WETLANDS

Given the overlap between waterfowl habitat and agriculture outlined in Chapter 3, as well as the importance of pesticides to western agriculture outlined in Chapter 4, we now wish to consider the possible contamination of waterfowl habitat with pesticides in regions where the overlap with chemical pest control is great. Contamination of this habitat can occur via direct overspray, drift deposition, vapour drift or surface and ground water runoff. A specific concern is the aerial application of pesticides in regions where the large numbers of productive sloughs make avoidance of direct application nearly impossible and downwind drift very likely. In these regions, most of the sloughs are small, slough distribution is random and the distance between adjacent water bodies is short (often <100 m). In addition, the size of the margin between cultivated land and slough water, which is often small, may be of little significance in preventing contamination if complete spray coverage of all pest refuges (including margins) is necessary to reduce pest numbers to acceptable levels. Also, sloughs, as catchment areas for drainage basins, may be subject to the accumulation of agricultural chemicals through runoff. This chapter examines the likelihood of wetland contamination by pesticide use in the Canadian prairies. Specific contributing factors examined include application strategies, application techniques, meteorological influences, topographical and geographical influences as well as compound specific properties affecting pesticide mobility. Some of these factors also have some bearing on the extent to which upland nesting areas are subject to drift or direct contamination resulting from both ground and aerial application of pesticides.

5.1 Pathway of Entry

The entry of pesticide contaminants into wetlands is either through direct application, indirect drift (droplet and vapour), runoff or seepage. The likelihood of direct contamination is highly dependent upon the selected mode of application whereas indirect contamination is dependent on meteorological events and specific pesticide properties. The method of application with the greatest probability of resulting in direct contamination is aerial application.

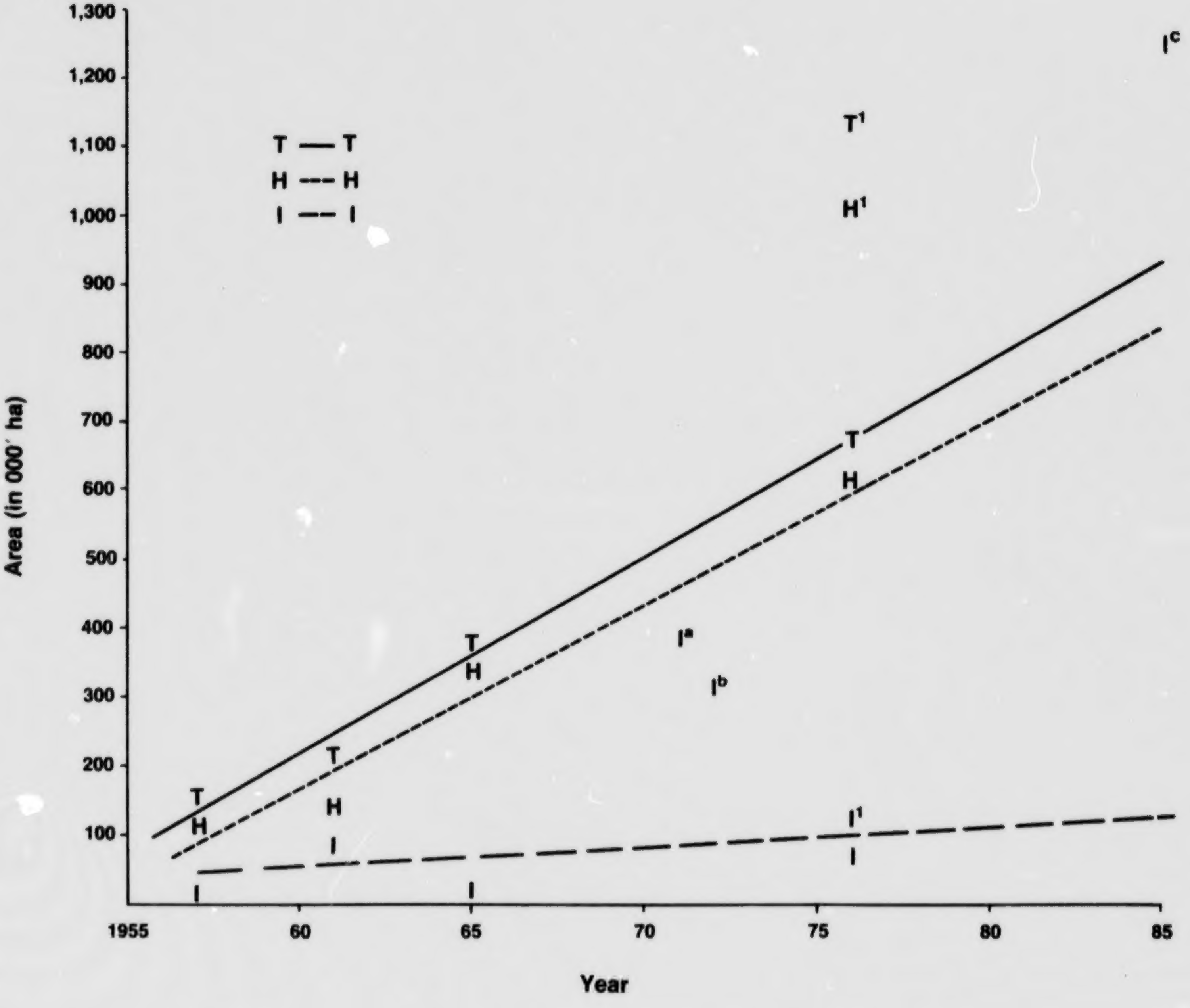
Accurate information on the importance of aerial application to total pesticide delivery in any one year is impossible to obtain. One of the reasons for this is the large number of commercial and private aerial applicators in western Canada. Three categories of applicators are recognized: commercial (licensed) operators, exemption operators (who are exempt from licensing as long as they operate within a 25-mile radius of their home base) and "flying farmers" operating privately registered aircraft over their own farms only (King 1978). In the three prairie provinces, it is estimated that, in 1976, 50% and 39%, respectively, of aerially applied herbicides and insecticides were applied by unlicensed operators.

The only recent information on the importance of aerial applications in Canada comes from King (op. cit.) who conducted comprehensive surveys of commercial, exemption and "flying farmer" applicators in 1957, 1961, 1965 and 1976. The total surface areas reported to have been sprayed aerially for weed and insect control are presented in Figure 5.1. Although sampling intensity was comparable in those four years, the return rate during the 1976 survey was probably low relative to the other years and, therefore, the area sprayed aerially in that year as well as the rate of increase of area sprayed were underestimated (G.J. King, pers. comm.). It was assumed in the surveys that all potential aerial applicators were contacted. This is probably not the case with "flying farmers" since they would not have been censused if they did not belong to one of the three provincial "flying farmer" associations.

We consider the data on herbicide spraying to be the most accurate, given the non-fluctuating nature of herbicide use (see Chapter 4). After correcting for the low return rate in the 1976 survey, King (1978) estimated that 1.01 million ha had been treated aerially with herbicides. Extrapolating to 1980, the year for which we have information on pesticide use from the 1981 Census of Canada, we estimate that 1.1 million ha were treated aerially for weed control. This represents approximately 9% of the estimated 12.6 million ha treated with herbicides (Chapter 4). There is evidence that the proportion of total herbicide use that is aerially applied has not changed dramatically over the last decade. Figure 5.1 shows an estimated increase of 70% in the area aerially treated with herbicides between 1970 and 1980. This increase can be explained entirely by the increase in total herbicide use over the same period (76% - Section 4.3.1).

The extent and popularity of the aerial application of insecticides is more difficult to determine. The extent to which aerial application is employed depends entirely on the status of the various "outbreak" pests. Bertha armyworm and, more recently, the wheat midge are pests requiring almost 100% aerial application. In years of heavy outbreaks (e.g. the bertha armyworm outbreak of 1971-72), Canadian aerial applicators were saturated and U.S. applicators were recruited for the spraying (King 1978). As far as we can determine, the year 1976 was not characterized by any major insecticide spray. Grasshopper populations were below normal (A. Ewen, pers. comm.) and, although a heavier-than-normal flea beetle infestation was reported (Table 4.5), only a small proportion of total insecticide use against flea beetles is in the form of foliar spray (Lamb & Turnock 1982). King (1978) estimated that the total aerial use of insecticide was 127,000 ha in that year. Given a "baseline" insecticide spray area of 1-1.5 million ha (Chapter 4), the 1976 aerial spray accounted for an estimated 8.5-12.7% of the total insecticide use. Historically, the area treated aerially for bertha armyworm has been almost as high as 400,000 ha. In 1985, an estimated 13% of all grasshopper control (circa 610,000 ha) and probably a similar proportion of cutworm control (circa 14,000 ha) were carried out aerially (Section 4.2). Given also the 150,000 ha which were aerially treated for wheat midge and another 480,000 ha treated for diamond-back moth in canola, the total aerial use of insecticides in 1985 covered approximately 1.24 million ha. This estimate is far above that obtained from the extrapolation of the surveys by King (1978), emphasizing the difficulty of predicting the extent of aerial application of insecticides.

Figure 5.1. Surface area estimated to have been sprayed aerially with pesticides in the three prairie provinces, 1957 to 1985. H, I and T refer to the area sprayed for herbicides, insecticides and both, respectively, combined on the basis of uncorrected survey returns by Agriculture Canada in 1957, 1961, 1965 and 1976 (King 1978). H^1, I^1 and T^1 are estimates corrected for non-returns of the 1976 survey. I^a and I^b represent the total bertha armyworm spray for 1971 and 1972; I^c was calculated assuming that 15% of the grasshopper and cutworm infestations and the entire wheat midge and diamond-back moth infestation were treated aerially in 1985 (see text).



Area (in 000' ha)
1,300
1,200
1,100
1,000
900
800
700
600
500
400
300
200
100
T — T
H --- H
I — — I
T
H
I
T
H
I
T
H
I
T
H
I
I^1
T^1
H^1
I^a
I^b
I^c
1955
60
65
70
75
80
85
Year

In the process of obtaining these data, we learned that Agriculture Canada has no intention of updating the work of King and his predecessors. This is unfortunate in view of the need for more accurate and up-to-date information on aerial spraying (Voisey 1984).

There are no hard data that would allow us to assess whether aerial spraying for insect control is increasing in popularity. We do have an indication of increased interest in aerial insecticide application from current research efforts into ultra low volume (ULV) spraying and from insistent requests by insecticide manufacturers that aerial delivery of new products be allowed. Certainly, the use of aircraft to apply pesticides in Canada is lagging far behind the use in the U.S. (Voisey 1984) and some increase in use appears certain. A further development is the recent introduction of ultralight airplanes modified for spraying crops. These are relatively inexpensive to purchase and the unit cost of spraying per acre using an ultralight is 7 to 10 times less than the cost using other planes (Anonymous 1984). The low cost makes this technology very attractive to farmers.

5.2 Control of Deposit

Drift is made up of the fraction of an applied pesticide that is deposited at variable distances downwind from the target area plus the fraction which remains airborne for an extended period of time. These contribute, respectively, to local and widespread "airmass" contamination problems (Robinson & Fox 1978). Of importance to waterfowl habitat are both aspects of herbicide drift: droplet deposition into margins and wetlands and airborne drift damage to emergent plants. However, only direct drift deposition of insecticides to slough water near crop targets is likely to affect aquatic animals. Drift is not limited to aircraft application; to a lesser extent, it is associated with groundrig application as well. The exception is ultra low volume (ULV) fogging by groundrigs. Maybank and colleagues (1976, 1978a, b) reported that groundrig application of 2,4-D formulations resulted in 3-5% downwind drift of initial droplets. In contrast, aerial spraying of 2,4-D contributed to off-target losses of 15 to 35% of applied amounts. Although aircraft spraying resulted in much higher initial drift, the resultant droplet cloud was coarse and tended to fall out more readily as it moved downwind than did the droplet cloud from groundrig spraying. For example, off-target glyphosate deposits at 30 m downwind were 3-30 times greater for aerial than for ground application of water diluted sprays (Yates et al. 1978). Multiple swath applications increase drift dramatically (Kaupke & Yates 1966).

Mist (ULV) application by both ground equipment and airplanes results in extensive downwind loss of pesticides. Impingement on off-target crops from groundrig mist blowers exceeded that from aerial application of larger droplet emulsions at all distances sampled downwind (Ware et al. 1969a). As expected, deposit and drift losses from an aerial ULV application approached 65% in comparison to 37% for aerial spray at higher volume of an an emulsion of fenvalerate (Ware et al. 1984).

It should be noted that there is much variability between studies estimating drift. This is due to the large number of factors affecting drift and to the variability associated with the different techniques used. Off-target drift from aerial spraying is highly variable and can be substantially greater than measured by Maybank et al. (1976, 1978a, b) and colleagues. In field studies of techniques to maximize 2,4-D low volatile ester applications on target areas, between 19 and 57% of the applied herbicide was deposited on target with the remainder accounted for by downwind drift droplet deposits and atmospheric vapour (Renne & Wolf 1979). In another study involving 16 separate applications of the insecticide methoxychlor, Ware et al. (1970) reported 14 to 95% deposition in the target area with an average of 54% drifting off-target. In two years of experiments with deltamethrin applications the amount of on-target deposit varied from 28% to 79% over ten replicates (Hill & Kinniburgh 1984). Studies where drift is sampled directly will be more reliable than when it is determined indirectly through deposit sampling (J. Maybank, pers. comm.).

5.2.1 Factors influencing pesticide drift

A number of factors influence particle deposition and drift. These were summarized in the 1960's by Akesson and Yates (1964) and more recently by Göhlich (1983). Basically, aerial drift is a function of droplet size and wind speed. The smaller the size of the droplets, a function of physico-chemical properties of the formulation and atomization processes, the longer their suspension in the atmosphere and the greater their lateral displacement due to wind. However, there is an inverse relationship between droplet size and application coverage. The effect of droplet size on horizontal drift and effective coverage is seen in Table 5.1. Obviously, the selection of appropriate droplet size spectra is a compromise between reducing drift (larger droplets) and maximizing coverage (smaller droplets). It is evident from many studies of pesticide drift that particles up to 50 µm will remain airborne for considerable distances. Thus, in order to ensure practical drift control, it is essential to eliminate, as much as possible, droplets smaller than 50 µm and perhaps 100 µm. However, to maintain effective spray coverage for insect pest control, the main body of droplets (the volume mean diameter [VMD]) should not exceed 250 µm. Table 5.2 presents a breakdown of droplet size distribution from various atomization and delivery sources. It is obvious that neither aerosols nor very coarse sprays present an acceptable droplet distribution.

Second to droplet size, the most important factor determining the amount of drift is aerial transport. Once the pesticide is released from the application vehicle it is subject to atmospheric variables which will determine the length of its stay in the air, its physical transformation during that time and the direction and speed of its movements. Tests which isolated the influence of wind speed on drift from aircraft application showed 84% on-target deposit at an average wind speed of 2.8 km h^{-1} and only 50% at 5.1 km h^{-1} (Ware et al. 1971). Similar trends with wind speed are reported for groundrig applications, although total on-target

Table 5.1. Effect of drop size on terminal velocity, drift distance and coverage [a].

Particle Diameter µm	Terminal Velocity[b] m/s	Horizontal Drift Distance[c] m	Drops/cm^2 Based on 10 l/ha
10	0.003	3704.0	191,000.0
50	0.076	146.0	1,528.0
100	0.27	41.0	191.0
200	0.76	14.6	23.9
300	1.15	9.7	7.1
400	1.62	6.9	3.0
500	2.07	5.4	1.5
1000	4.02	2.8	0.2

[a] From Yates *et al*. 1981

[b] Sp. grav. = 1.0

[c] Released at 5 m elevation, 8 km/hr wind

Table 5.2. Drop size distribution in aerosols and sprays (numbers are cumulative percent by volume)[a].

Drop Size µm	<50 µm Fine Aerosols	50-100 µm Coarse Aerosols	100-250 µm Fine Sprays	250-400 µm Medium Sprays	400-500 µm Coarse Sprays	500 µm Very Coarse Sprays
1-5	5	0.1				
5-10	45	0.4	0.1			
11 µm VMD[b]	50%					
10-15	77	2.0				
15-20	97		2.0			
20-40	100	12.0		0.1		
40-60		35.0	5.0	2.0	0.01	0.001
60-80						
86 µm VMD		50%				
80-100		59.0	15.8	6.0	0.1	
100-120						
120-140					0.4	0.1
130 µm VMD			50%			
140-180	100.0					
180-200			81.0	17.0	3.0	
200-220						
220-240					7.0	5.0
240-260						
260-280						
280-300			100.0	46.0	14.0	
278 µm VMD				50%		
300-350					24.0	
350-400					36.0	15.0
400-450					46.0	
460 µm VMD					50%	
450-500				92.0	55.0	
500-600					74.0	25.0
600-700					88.0	
700-800					96.0	
900 µm VMD						50%
800-1000				100.0	100.0	100.0
VMD	11 µm	86 µm	130 µm	278 µm	460 µm	900 µm
Nozzle Type	Cold Fogger 0.35 kg/cm^2 Air	2-fluid 2.1 kg/cm^2 Air No air-stream	Spinner 140-160 kph Air	65015 Fan Down 2.8 kg/cm^2 140-160 kph Air	D6-46 Cone Back 3.5 kg/cm^2 140-160 kph Air	D6-Jet Back 2.8 kg/cm^2 140-160 kph Air

[a] From Wilce et al. 1974

[b] The VMD (volume median diameter) is that size of drop for which 50% of the volume is in drops above that size and 50% are below the VMD size.

deposit is greater. Both the initial off-swath deposit and vapour drift increased with increasing wind speed with ground delivery of 2,4-D amine solution in 7 to 26 km h^{-1} winds (Maybank et al. 1977).

Atmospheric stability is also critical in determining the amount of off-target drift that will occur. During conditions of inversion (i.e. the air is colder near the ground and warmer above), the stable air mass maintains a shallow layer of spray and gives rise to vapour drift near the ground which may drift horizontally for long distances without dispersing. In contrast, under lapse conditions (i.e. the warmest air is near the ground and rises thereby creating eddies and turbulence), the amount of spray reaching the target is lower but the drift is spread through a greater depth of the atmosphere and therefore may not cause much off-target damage. In one study (Yates et al. 1966), deposits 305 m downwind from a target strip were ten times greater under strong inversion conditions than under very unstable conditions.

Specific factors contributing to an increase in drift potential can be categorized as follows:

Properties of the final tank mixture

(1) decreases in surface tension result in reduced droplet size

(2) reductions in viscosity decrease droplet size

(3) highly volatile tank mixes result in rapid evaporation of the solvent fraction of individual falling droplets

Application equipment/practices

(4) atomizers and atomization practices producing a high proportion of small droplets released at high pressure (e.g. jet air streams)

(5) increased release height

(6) nozzle orientation encouraging rapid horizontal or vertical upward movement of the spray

(7) airplane type, "angle of attack", speed and flight line producing high wind turbulence and droplet shatter or erratic placement of the swath or control of its size

(8) increasing the number of swaths

(9) operator control problems (e.g. failure to close nozzles at edge of field)

Meteorological

(10) wind direction and speed favoring off-target transport

(11) a. low relative humidity and high temperature enhance evaporation of the water fraction of the final tank mixture thereby decreasing droplet size

b. high temperature can also affect evaporation of other components of the tank mixture

(12) inversion conditions

Topological

(13) rolling or rough terrain interfere with even pesticide delivery from both ground and aircraft

(14) proximity to sensitive target habitat accentuates the need for deposition control

5.2.2 Factors specific to prairie agriculture

Meteorological and topographical features are expected to have major regional influences on off-target drift of pesticide sprays. Although formulation properties and regional application practices generally do not differ widely, some comments with regard to spraying prairie crops are offered.

5.2.2.1 Formulation and final tank mix

Specific information on actual adjuvant and carrier additions to pesticide formulations is generally not known for most agricultural situations. These solvent additives, to a large degree, determine the viscosity, surface tension and possibly the volatility of the applied formulation and thus may affect drift. Widely used emulsifiable concentrates are often 50% or more petroleum derivates. However, the influence of solvent composition on drift has not been adequately assessed. Argauer et al. (1968) reported that recovery of the emulsifiable concentrate of malathion and azinphos-methyl was greater within the target area and also at 30 m downwind from the target swath than for applications of the technical material without solvents. The composition of the adjuvants is at times changed by the manufacturer and may also be modified by the applicator.

The specific formulation properties of 2,4-D are particularly relevant to the Canadian prairies where it is widely used for broadleaf weed control. Approximately 35% of 2,4-D butyl esters, 12% of 180-octyl formulations and 5% of the amine formulation evaporated from the ground and migrated downwind after application to Saskatchewan fields (Grover et al. 1972, 1985, Maybank et al. 1978b). Based on quantities applied and formulation alone, 2,4-D esters are expected to be the largest contributors to off-target contamination due to vapour drift.

5.2.2.2 Application practices

In regions of high pond density, problems of application control will be paramount to wetland contamination. This aspect will be developed in Section 5.3.

5.2.2.3 Meteorological

High wind, high temperature and low relative humidity contribute to increased pesticide drift. The moderate average temperatures (18°-20°C) and precipitation (5.0-8.0 cm) in the prairies during the months of June and July in normal climatic years (Weir 1971) suggest relatively suitable spray conditions. However, wind in the prairies is a major destabilizing factor. Maybank and Yoshida (1969) concluded that winds in excess of the 16 km per hour produce significant pesticide spray drift that is not localized to adjacent fields but would instead contribute to general contamination. They indicate that excessive wind conditions occur near Regina and Saskatoon for 10-15 days of June. Millar (1969) indicated that the average wind speed for several wetland areas in Saskatchewan was in excess of the 16 km h^{-1} limit throughout much of the year. Therefore wind speed is expected to be one of the major factors influencing the amount of on-target deposit and drift of both groundrig and aerially applied pesticides in the pothole habitat.

Adverse spray conditions are also associated with drought (low relative humidity) and with summer temperature inversions. Drought conditions have occurred in the prairies in 12 of the past 35 years and inversions are a common diurnal feature in the prairies in summer.

5.2.2.4 Topographical

Although the Canadian prairies are often envisioned as flat, the land is rolling in much of the prairie pothole region. Local deviations from flat terrain and local obstructions such as potholes, shrubs and trees contribute to problems in spray control. Terrain roughness forces the farmer to slow his tractor thereby increasing spray drift, and also causes wheel bounce further increasing off-target drift (J. Maybank, pers. comm.). The contributions of topographical irregularities to spray drift should be greatest in the aspen parkland region where ponds and natural vertical obstructions are most abundant. This region also supports the greatest density of breeding ducks (Chapter 3).

5.3 Analysis of Potential Slough Contamination from Direct Sources

The difficulty of controlling pesticide deposition within the target area is reasonably well documented. However, there has been no effort to experimentally measure overspray and drift deposits in regions where potholes are abundant and in close proximity to spray operations. Our analysis of the likelihood of slough contamination from pesticide applications is therefore based on data indicating the lack of control over deposition using both aerial and groundrig application techniques and data suggesting the probability of some direct overspray. Major sources of contamination are expected to be direct overspray and drift droplet deposition from target swaths close to the slough and margin vegetation.

5.3.1 Overspray

The most conspicuous pathway to wetland contamination in the Canadian prairies pothole region is direct aerial overspray. However, the probable frequency of direct overspray and the likely inputs from this source are difficult to assess.

No direct studies of the overspray problem have been undertaken, nor are there drift data from regions with extensive numbers of potholes. Analysis of the likelihood of direct overspray in such pothole regions depends on assumptions about applicator philosophy and ability, specific limitations in aircraft control and the size and dispersal pattern of wetlands. An applicator's desire for complete coverage and his perception of sloughs as something requiring protection from pesticide contamination or as pest refuges determine, in part, if sloughs are purposely oversprayed. In cases such as grasshopper infestations, application practices dictate the spraying of all possible refuges. For small sloughs (<0.4 ha) with narrow margins common in cultivated lands, partial overspray is highly probable. These ponds may account for 60 to 80% of the wetland habitat available to ducks in May-June (see Chapter 2) when both insecticide and herbicide spraying are likely. If the applicator does not perceive a slough as habitat to be protected, small to moderately-sized sloughs will most certainly be overflown to assure complete coverage with minimal labor and cost. The perception of sloughs by farmers is evident in statements made by a representative of prairie farmers at a recent conference on the "Future Role of Aviation in Agriculture".

> "As for coverage, all sloughs, fencelines, rock piles and road allowances are sprayed where wheeled equipment cannot go. This applies to herbicides as well as dangerous insecticides ... In closing, I mentioned sloughs, and in my farm operation a slough is a portion of my field that is seeded and the next day gets flooded out, covered with ducks spreading seeds from Canada thistle or Sow thistle, etc., into the slough area" (Peterson 1984).

Similarly, a farmer-applicator at the same conference said the following about sloughs:

> "... field sprayers ... also have to work around or through our sloughs. To work around a slough with a field sprayer you must either overlap or miss while the aircraft can proceed across the slough" (Gillis 1984).

Clearly, farmers and applicators consider dugouts and large permanent ponds as aquatic bodies deserving protection but regard the temporary or semi-temporary sloughs (which are in fact the most important to waterfowl in wet years) as pest refuges even when "covered with ducks". Official advice from provincial authorities is no more reassuring. The following is taken from the annual "Grasshopper Control" publication of Saskatchewan Agriculture:

> "The most important areas to inspect [for grasshopper] are ... 5) wasteland areas such as drift soil along headlands, field margins, sloughs and fencelines ... The earlier the grasshoppers are controlled [with chemicals] on the egg bed areas in pastures, roadside ditches, sloughs and headlands, the better."

Even in such instances when wetland protection is desired by the applicator, the narrow margins (F10 m) as well as small size and close proximity of these water bodies to cropland may make avoidance of direct overspray nearly impossible. Applicators have severe restrictions in airplane maneuverabilty and control of release at speeds in excess of 130 km h^{-1} and swath widths of 10-20 m. At 150 km h^{-1} an airplane would take approximately two seconds to fly between two small sloughs (50 m diameter) separated by 100 m of cropland and less than one second to fly over the individual potholes. No applicator could be expected to suitably apply the pesticide without direct contamination of the two sloughs. Analyses in Shaw et al. (1984) show that in some prairie regions (e.g. Alberta, Provincial Section 29) slough density approaches 10 potholes km^2 and the average distance between adjacent potholes, all of which were less than 4092 m^2 in area, was approximately 185 m. Average pond density estimates from waterfowl surveys (see Section 3.2.4) were 7.8/km^2 in parklands, 5.8/km^2 in mixed prairies and 2.8/km^2 in shortgrass prairies. In intensively studied waterfowl areas in the aspen parklands, more accurate estimates of pothole densities were 2-3 times higher than census averages. Smith (1971) reported a mean of 16.4 potholes per km^2 at the Lousana, Alberta, site for the 13-year study period. These ponds accounted for approximately 55 ha of surface water in June or about 5.5% of the total average. The figures for Redvers, Saskatchewan, were similar with a mean of 19.7 potholes per km^2 on June 1 covering approximately 65 ha or 5% of the land surface (Stoudt 1971). Although neither Smith (1971) nor Stoudt (1971) presented data on spacing between ponds, temporary and seasonal potholes of 0.4 ha in size made up over two-thirds of the total ponds in these parkland habitats. Aerial application in such regions would certainly result in contamination from both direct overspray and drift deposition. Larger sloughs are less likely to be contaminated.

In the worst case of direct aerial overspray, 100% of the applied amount is assumed to be deposited on ponds and margins. This "worst case" assumption provides a basis for compound comparisons as seen in hazard models in Section 6.3 but is not valid for actual application conditions. Under a variety of meteorological and application conditions, on-target deposits have been found to vary substantially and rarely approach 100%. In fact, an average on-target deposit of approximately 50% of applied quantities from a single swath may be realistic (Ware et al. 1970) and can be assumed to fall on the surface of an oversprayed slough. Each additional complete overflight of a slough increases contamination. Overspray deposits of 80% of applied quantities are expected from multiswath coverage which is common in insect control (N.B. Akesson, pers. comm.).

While direct overspray of sloughs with groundrigs is less conspicuous than with airplanes, this pathway of contamination may be quite important in light of the dominance of groundrigs in the application of pesticides in the prairies (see Section 5.1). The probability of direct overspray is greater in insect control than in weed control because, in the former, the operator wants to spray as much of the uncultivated slough margins as possible and probably allows the boom to swing directly over the pond. The likelihood of direct overspray when applying herbicides depends on the size of the margin between the slough and the field.

5.3.2 Drift droplet deposition

The second substantial source of pesticide contamination is drift droplet deposition from upwind applications. Control of pesticide deposition at distances closer than 10-15 m from the swath would appear to be extremely difficult. For example, in two trials, an average of 6.1 and 8% of applied 2,4-D was measured in the initial off-swath deposit in a 5-metre wide strip downwind (Maybank et al. 1978a). Two to three times these amounts were still airborne at a sampling line 5 metres downwind. Currier et al. (1982) found significant carbaryl deposition 12 metres downwind and upwind of a sprayed orchard, regardless of flight pattern and prevailing wind. Under conditions of negligible wind (3.5 km h^{-1}) and a slight inversion, an estimated 12-19% (2.8-4.2 g cm^{-2}) of the applied carbaryl was deposited at sampling sites 12 meters from the target. These results show that even under semi-optimum application conditions (low wind, narrow spectrum of large droplets) substantial contamination is probable close to the swath and cannot be prevented. The deposits may be caused in part by the air turbulence in the wake of the passing aircraft.

The concentration of off-target pesticide deposited from aerial spray at distances greater than 5 metres decreases rapidly with distance (Maybank et al. 1978a). The studies cited in Table 5.3 show that deposits, estimated as a proportion of the application rate, were between 0.1 and <12% at 50 m and generally less than 1% at 100 m downwind; at 500 m downwind from the swath, <0.1% ofapplied quantities was detected. The carbaryl study by Currier et al. (1982), where off-target concentrations were measured radially from the target centre, suggests that the positioning of sampling stations in some earlier studies may have caused an underestimate of deposits. A summary of pesticide recovery with distance downwind indicated that between 15 and 40% of the applied amount will fall between 10 and 100 m downwind of the plane flight line (Akesson and Yates 1984). Sprays with droplets of 150 to 450 μm volume mean diameter (VMD), the normal range for most aerial applications, had greater than 20% of applied amounts fall in the 10-200 m off-target range (Ibid). Downwind areas at distances greater than approximately 10 m may receive substantial inputs under conditions conducive to drift, particularly from sprays with a small average droplet size. The data suggest that drift droplet deposit could supply an

Table 5.3. Off-swath deposit of aerially applied pesticides.

		Deposit as Percent of Application Rate				
Pesticide	Application Data	50 m downwind	100 m downwind	500 m downwind	Comments	Reference
malathion	Piper monoplane; D6-45 hollow core nozzles straight down; 160 km h^{-1} flight speed	6.5	1.5	0.04	applied under turbulent conditions	Argauer et al. 1968
azinphos methyl	Piper monoplane; D6-45 hollow core nozzles straight down; 160 km h^{-1} flight speed	10.5	0.9	0.2	applied under turbulent conditions	Argauer et al. 1968
methoxychlor	Piper monoplane; 56-4664 Tee Jet; D8 nozzles at 345 kPa; 129 km h^{-1} flight speed; 15 m swath; 4.8-8.1 km h^{-1} wind speed	1.2	0.9	0.09	1.2% of concentration on target found 800 m downwind	Ware et al. 1969a
methoxychlor	Stearman biplane; no. 8-36 nozzles at 207 kPa; 125 km h^{-1} flight speed; 12.8 m swath; 1.6-4.8 km h^{-1} wind speed	1.1	0.9	0.1	less drift during morning application	Ware et al. 1969b
fenvalerate	Piper monoplane; D7-46 nozzles at 145 kPa; 161 km h^{-1} flight speed; 8.7 km h^{-1} wind speed	-	0.5	0.15		Ware et al. 1984
carbaryl	Piper monoplane; no. 6-36 flat nozzles at 3.52 kg cm^{-2}; 144-177 km h^{-1} flight speed; 16.8 m swath; 0.3-13.0 km h^{-1} wind speed	0.7-12.0 (76 m)	0.6-1.5 (150 m)	-	greater drift under temperature inversion	Currier et al. 1982
glyphosate	Piper monoplane; D6-46 nozzles at 176 kPa; 160 km h^{-1} flight speed; 10 m swath; 12.4-15 km h^{-1} wind speed	0.1	0.02	0.005		Yates et al. 1978
2,4-D (BE and PGBE)	Piper monoplane, D8-38 Tee Jet nozzles at 2.7×10^{-5} bar; 144 kmh^{-1} flight speed, 16m swath; RH = 37 to 74% on different runs	0.8-2.0	0.5-1.4	-	greater drift at low relative humidity	Renne and Wolf 1979

additional 10% or more of applied amounts to sloughs within 50 m downwind of the swath. Approximately 1% of applied amounts would be expected to further contaminate sloughs 100 m downwind of spray swaths.

Downwind sites are likely to accumulate pesticides gradually from each swath applied. High residue concentrations would occur primarily from a large number of upwind swath applications in addition to direct overflight. This situation could produce acute exposure of sensitive off-target plants and animals to pesticides. Reports of crop damage from herbicides and invertebrate kills from insecticides at substantial distances downwind indicate both acute effects and longer term chronic effects are possible. For example, fixed-wing application of the herbicide glyphosate, at 4.2 kg ha^{-1} for a 100-meter wide treatment, produced 30% or greater visible injury to four-leaf wheat plants up to approximately 160 m downwind (Yates et al. 1978). A buffer zone approximately 300 m wide was required to prevent 30% visible injury to grape plants under the same application conditions. Herbicide impact on margin and emergent wetland plants as a consequence of spray drift from aerial applications is not well documented. Drift of endrin and permethrin aerially applied to crops for cutworm control was sufficient to kill, within 24 hours, all Daphnia magna exposed in beakers of water at distances up to 360 m downwind (Schladweiler & Weigand 1983). The preceding analysis of drift droplet deposition resulting from aerial application of pesticides applies equally as well to groundrig application. Mechanical turbulence and release height are minor factors affecting drift when compared to droplet size, wind speed and atmospheric stability (J. Maybank, pers. comm.).

As previously described (see Section 5.2.2.4), ground contours and high terrain roughness around slough borders often result in increased off-target drift. This increases the chance of contamination not only of sloughs, but also of areas providing the best nesting cover for waterfowl in an agricultural landscape: the slough margins and upland habitats such as native grasslands, edgerows, ditchbanks, etc. This is of particular concern when farmers spray herbicides up to the edge of their fields where those compounds can affect vegetative cover or spray acutely toxic pesticides (e.g. carbofuran) which contaminate the nesting habitat of the birds.

The above data analysis strongly suggests that no form of mitigation such as buffer zones is feasible to control off-target pesticide deposition in regions of high pothole density when compounds are applied aerially. While this method of application is minor when compared to the use of groundrigs in the Canadian prairies, the latter method is no less a source of contamination for both slough and waterfowl nesting cover.

5.4 Contamination from Indirect Sources

Although direct wetland contamination from off-target application is a major exposure vector, highly volatile and surface mobile pesticides may enter the wetland indirectly from cropland.

5.4.1 Vapour drift

Monitoring of airborne residues, both in the Canadian prairies and elsewhere, has shown that post-application losses of volatile herbicides can be an important mechanism of contamination outside the target area. Volatilization losses of 30-40% and 10-15% have been reported for the butyl and iso-octyl esters, respectively, of 2,4-D (Grover et al. 1972; Maybank et al. 1978b). In a later report Grover et al. (1985) estimated cumulative vapour losses of the iso-octyl ester of 2,4-D to be near 21%. Similar vapour drift values from 2,4-D formulations were estimated by Gile (1983) who reported that 5.1% of applied BEE formulation, 8.5% of applied PGBE formulation and 15% of butyl and iso-octyl esters were lost from a simulated wheat field. Post application vapour losses of trifluralin was reported to be 25% of applied quantities (White et al. 1977). EPTC vapour losses are as high as 75% of applied herbicide concentrations (Cliath et al. 1980). The highest concentration of vapour were in samples nearest to ground level (Grover et al. 1985).

As the 2,4-D esters (BEE and PGBE) are widely applied throughout the prairie region as is trifluralin in areas of oil-seed crops (see Chapter 4), vapour drift could be an important vector of wetland contamination. However, probably only emergent plants are at substantial risk of vapour exposure and the low-lying topography of wetlands may minimize their exposure. Emergent aquatic macrophyte exposure to vapour drift of highly volatile herbicides is roughly estimated as 1-10% of applied concentrations.

5.4.2 Surface runoff

Slough contamination via soil leaching and surface runoff from cropland is minimal for most hydrophobic pesticides due to adsorption phenomena. For example, the synthetic pyrethroid compounds show minimal lateral surface movement in contaminated soils. Deltamethrin, cypermethrin and permethrin were found to be immobile in both soil column and soil thin-layer chromatographic systems (Kaufman et al. 1981). Fenvalerate was not readily leached (<0.1%) in microplot field studies with total rainfall of 354 mm including a 26 mm downpour 15 days after application (Hill 1981). Under conditions of high rainfall, Smith et al. (1983) reported 0.56% of the applied fenvalerate in runoff from sugarcane plots. In full-scale field tests, residues of permethrin in run-off from cotton fields did not exceed 0.20 ppb and the total recovered in water + sediment was <0.10% of the amount applied (Carroll et. al. 1981). In abnormally high rainfall conditions <1% of the applied amount was recovered in runoff. Much of the runoff loss of the pyrethroids is probably associated with particulate erosion.

Although the relationship between water solubility and leaching is not simple because soil sorption of pesticides can be related to both surface charge attraction and the attraction of non-polar compounds for soil organic materials, more soluble pesticides generally more readily leached.

Approximately 90% of carbofuran, 20% of carbaryl, 7% of diazinon and 3% of parathion were leached from organic soil in two successive rinses under laboratory conditions (Sharom et al. 1980a). In contrast, less than 2% of chlorpyrifos and organochlorine compounds were leached in 10 successive rinses in the same study. Because of their high water solubility, dimethoate, malathion and terbufos are thought to be highly mobile in soil runoff (Bowman & Sans 1982). However, methoxychlor is not readily leached (Ibid). Azinphos methyl was leached from sugarcane plots at levels of 0.08% and 0.055% of applied amounts during, respectively, normal and heavy precipitation years; leaching of fenvalerate was comparable (Smith et al. 1983).

Many of the more widely used herbicides in the Canadian prairies are highly mobile in soil solution and possess a substantial degree of residual phytotoxicity. Included in this group are the amine salt of 2,4-D, dicamba, amitrole, dalapon and a number of other highly water-soluble compounds. However, the most widely used herbicide, triallate, does not fall into this category. During spring runoff, the loss of fall-applied 2,4-D to grain stubble in Saskatchewan was 4.1% of the amount applied (Nicholaichuk & Grover 1983) and this amount is thought to be similar to losses expected with runoff after a severe storm. The average flow-weighted mean concentration for snowmelt runoff was 31 μg L^{-1} (Ibid). Mayeux et al. (1984) reported that about 6% of applied picloram left the treated area during one month when conditions were especially conducive to surface water runoff. Maximum concentrations of picloram were 48-250 μg L^{-1} in runoff water and concentrations decreased with each successive runoff event.

In summarizing the seasonal losses of pesticides in surface water from agricultural fields (Table 5.4), Wauchope (1978) concluded that, for the majority of commercial pesticides, runoff losses were 0.5% or less of the amount applied unless severe rainfall events occur within 1-2 weeks after application. Total runoff loss is generally less than 10 percent of quantities applied. Higher losses have been measured for paraquat with most of the herbicide associated with crude soil particles (Smith et al. 1978). Atrazine, cyanazine and 2,4-D compounds show a consistent loss in watershed runoff of 2-8% of applied amounts. Wauchope and Leonard (1980) classified the wettable powders of triazine herbicides and the aqueous solutions of paraquat, 2,4-D, dicamba and picloram to be highly available in surface runoff. Under conditions of high rainfall shortly after pesticide application, runoff loss may approach 20% of applied amounts for highly mobile compounds (Table 5.5). Losses under "worst case" conditions are high for atrazine, cyanazine, 2,4-D esters and paraquat. Wauchope (1978) concluded that there is no reason to suppose that single-event runoff losses of 10% of any unincorporated pesticide cannot occur. Although the probability of such an event is quite small, this 10% runoff loss figure can be considered adequate in estimating maximum expected loading from surface runoff. Aqueous solutions applied directly to foliage are more mobile than those applied to bare soil. Granular formulations, emulsions and soil-incorporated compounds are less available in runoff.

Table 5.4. Seasonal losses of pesticides in agricultural fields due to surface runoff[a].

Pesticide	Total Runoff Loss (% of applied)	Reference
atrazine	1.7-5.1	Hall et al. 1972, Hall 1974
carbaryl	0.15	Caro et al. 1974
carbofuran	0.7-4.1	Caro et al. 1973
cyanazine	2.4-8.0	Smith et al. 1978
2,4-D (salt)	3.3-8.2	White et al. 1976
2,4-D (acid)	2.4-8.0	Smith et al. 1978
glyphosate	0.5-1.8	Edwards et al. 1980
linuron	0.9-3.2	Willis et al. 1975
methoxychlor	2.8	Edwards & Glass 1971
metribuzin	0.9-2.1	Wauchope et al. 1977
paraquat	0.9-18.3	Smith et al. 1978
picloram	0.002-4.5	Scifres et al. 1977, Davis & Ingebo 1973
simazine	3.5	Edwards 1972
trifluralin	0.001-0.8	Sheets et al. 1972, Willis et al. 1975

[a] Specific site, application and runoff conditions, except for glyphosate, summarized in Wauchope 1978

Table 5.5. Herbicide runoff from agricultural fields under "worst case" conditions of high rainfall shortly after application[a].

Pesticide	Total Runoff Loss (% of applied)	Reference
atrazine	5-17	White *et al*. 1967
cyanazine	4-19	Baker *et al*. 1976
2,4-D (acid)	2-10	Asmussen *et al*. 1977
2,4-D (salt)	3-8	Barnett *et al*. 1967
2,4-D (esters)	11-27	Barnett *et al*. 1967
dicamba	3	Trichell *et al*. 1968
glyphosate	2	Edwards *et al*. 1980
paraquat	5-18	Smith *et al*. 1978
picloram	3-6	Trichell *et al*. 1968

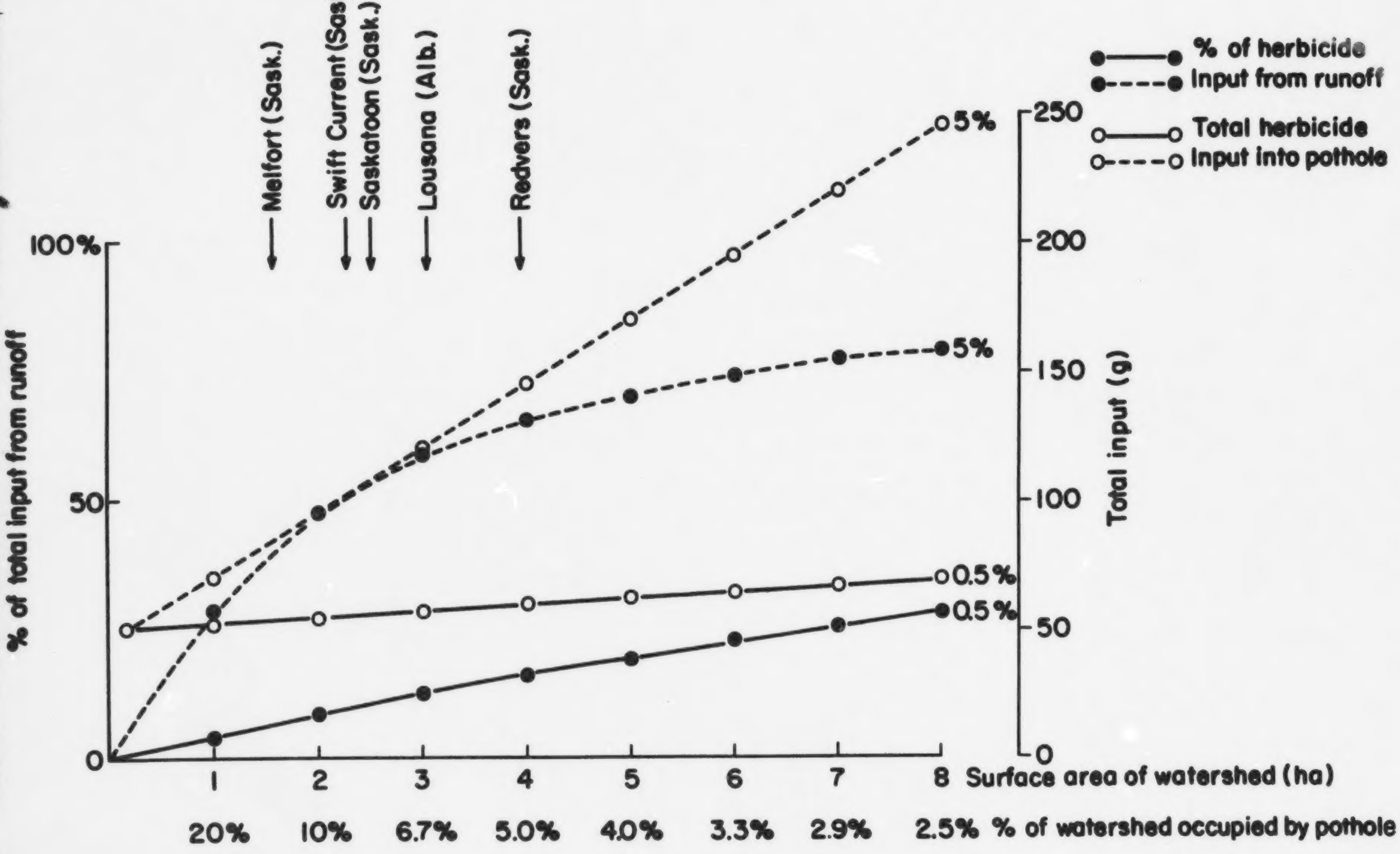
[a] Specific rainfall, runoff and application conditions for studies cited in this table are summarized in Wauchope 1978

The contribution of runoff to total pesticide loading in sloughs is a function not only of the mobility of the compound and the amount of aerial deposition (overspray and drift), but also of geographical factors such as the slope and size of the watershed. The relative importance of runoff input into sloughs as a function of watershed size is examined using a simple model. The herbicide chosen is 2,4-D (500 g AI L^{-1}) aerially applied at a rate of one L ha^{-1} over a watershed ranging in size from 0 to 8 hectares and containing a pothole 0.2 ha in area. This last figure corresponds approximately to the average pothole size in three study areas in Saskatchewan (Millar 1969). It is also assumed that 50% of the herbicide applied reaches the ground (see Section 5.3.1) and drift is ignored. The total input of herbicide into the slough is simply the sum of that amount reaching the water surface through direct overspray and of the herbicide runoff into the pothole. Two sets of calculations were done using two rates of runoff input: 0.5 and 5% of the amount reaching the ground. Figure 5.2 illustrates the effect of watershed size on total herbicide input and the contribution of runoff to total input. While the total input increased very little with area of watershed at runoff rates of 0.5, it did increase rapidly at rates of 5%. For ratios of pothole area to watershed area characteristic of the Canadian prairies (0.05 to 0.15; calculated from data in Millar 1969, Smith 1971, Stoudt 1971), herbicide loading into a slough can double with a tenfold increase in runoff rate. Figure 5.2 also clearly indicates that increases in runoff rates lead to increases in the contribution of runoff to total input. Again, in a situation characteristic of the Canadian prairies, runoff can contribute up to 50% of total loading during "catastrophic" runoff events.

While the preceding exercise demonstrates the role of weather and topography in determining the importance of runoff inputs, "catastrophic events" as described therein are rare and very localized and usually occur during the summer when crop cover is sufficient to reduce surface runoff. Such excessive rains also change the water regime and distribution of potholes, creating new areas of standing water and temporary sloughs because drainage is poor in the prairie pothole region (J. Maybank, pers. comm.). Furthermore, 85% of annual runoff is derived from spring snowmelt before pesticides are applied (W. Nicholaichuk, pers. comm.). A potentially greater source of concern is the leaching of pesticides into groundwater followed by discharge into sloughs. Groundwater flow systems are important to pothole hydrology and patterns of flow which lead to the buildup of dissolved salts could also lead to increased concentrations of contaminants. The extent of this problem has not been investigated.

Although the increased interest in no-till agriculture may mean greater use of mobile herbicides, studies indicate that runoff losses of herbicide from no-till agriculture are less than from conventionally plowed fields. The mulch cover in no-tillage increases infiltration and slows overland flow, thus decreasing total runoff loss of herbicides. Triplett *et al*. (1978) reported no detectable runoff of atrazine or simazine from seven no-till study plots and <1% during the season from five of seven of the remaining uncultivated plots; only three of nine plowed fields had no detectable runoff.

Figure 5.2. Contribution of runoff to total herbicide input to potholes. Solid lines assume runoff losses to be 0.5% of amounts applied while dotted lines assume losses of 5%. The five sites shown on the Figure indicate where those areas are located on the continuum with respect to the percentage of watershed occupied by a pothole.


Melfort (Sask.)
Swift Current (Sask.)
Saskatoon (Sask.)
Lousana (Alb.)
Redvers (Sask.)
% of herbicide
Input from runoff
Total herbicide
Input into pothole
% of total input from runoff
100%
50
0
Total input (g)
250
200
150
100
50
0
5%
5%
0.5%
0.5%
1
2
3
4
5
6
7
8
Surface area of watershed (ha)
20%
10%
6.7%
5.0%
4.0%
3.3%
2.9%
2.5%
% of watershed occupied by pothole

There are few monitoring studies of pesticide residues in sloughs. Gummer (1979) reported that runoff of the herbicide 2,4-D led to detectable levels (up to .33 μg L^{-1}) in prairie waters. Water pumped from the Holland Marsh drainage (2914 ha), an Ontario agricultural region, in 1974, had a total of 600 ng L^{-1} of organophosphorous insecticides and approximately 100 ng L^{-1} of organochlorine insecticides (Miles & Harris 1978). Despite the fact that persistent organochlorine compounds were not in wide use at that time, <100 g/week DDT and dieldrin were pumped from the marsh in water. This represents a persistent year-round chronic source of input. Diazinon was the only organophosphorous compound to exceed the <100 g/week level. Its input was more seasonal. In southern Ontario, atrazine is frequently found in stream waters in agricultural watersheds. Frank and Sirons (1979) detected atrazine in 80% of streams sampled at a mean concentration of 1.4 μg L^{-1}.

Models have been developed to predict pesticide runoff from agricultural land (see Crawford & Donigian 1974, Adams & Furisu 1976, Onishi et al. 1982). Processes and factors which determine the concentrations of pesticides available for runoff from agricultural land are discussed in detail by Bailey et al. (1974). Current models are equipped to simulate rainfall events, erosion, particulate and soluble transport and degradation kinetics. Unfortunately, there have been few reports of model application or of convergence between model predictions and field data.

5.5 Likelihood Summary

The importance of runoff to insecticide contamination of prairie sloughs is unknown but thought to be minimal in comparison to direct and drift deposits. Under conditions of high rainfall following application, runoff of soluble herbicides may be equal in importance to drift deposition in contributing to loading of the slough system. However, these events are rare and localized. Runoff events not temporally close to application will generally carry much smaller quantities of these herbicides and this loading vector will not be directly additive with those of overspray and drift unless the compound is highly persistent.

A best estimate of the likelihood of pesticide entry into sloughs in the pothole region is 50% of applied amounts via aerial (ground or aerial application) overspray, 1-10% additional due to drift deposition and generally <1 from groundwater, runoff and vapour drift.

CHAPTER 6
TOXICITY OF INSECTICIDES IN SLOUGH ECOSYSTEMS: INDIRECT THREAT TO WATERFOWL

This chapter examines the laboratory toxicity data for the synthetic pyrethroids and compares them, where possible, to the effects of other major use insecticides on aquatic invertebrates. Long-term impacts of insecticide-induced changes in experimental field studies are summarized with respect to structural and functional changes in the ecosystems, stability and resilience of the invertebrate communities and the potential influence of these factors in contributing, indirectly through reducing available food, to decreased waterfowl production. A systematic approach to the estimation of ecological hazard is described and a computer model is applied to predict the relative risk of an indirect threat to waterfowl from overspraying a shallow slough with selected insecticides including the pyrethroids.

6.1 The Comparative Toxicity of Pyrethroid and Other Major Use Insecticides to Aquatic Organisms

The reported toxicity values for fish and aquatic invertebrates exposed to the synthetic pyrethroids cypermethrin, deltamethrin, fenvalerate and permethrin are, in general, as great or greater than those for the most toxic organochlorine, organophosphorous and carbamate insecticides. The reported toxicity values for these pyrethroids are less than 10 µg AI L^{-1} and, in a number of cases, values are at or below 1 µg AI L^{-1} (see review in NRCC 1986 for data summaries on cypermethrin, fenvalerate and permethrin). The toxicity of these compounds to aquatic species varies inversely with temperature (higher temperature - lower LC_{50}), but is not affected by pH or water hardness within normal environmental ranges. The influence of suspended solids in reducing the toxicity of the synthetic pyrethroids is substantial. Formulated products were shown to be more toxic to rainbow trout than technical materials (Coats & O'Donnell-Jeffery 1979, Zitko et al. 1979). Another confusing factor is the toxic contribution of various isomers making up the insecticide mixture: the (+)-isomers (Miyamoto 1976) and the IR CIS isomer (Zitko et al. 1979) of permethrin are more lethal to fish than permethrin itself (a mixture of isomers). Degradation products are, in general, several orders of magnitude less toxic than the parent insecticide.

6.1.1 Aquatic insects

The synthetic pyrethroids are highly toxic to a wide variety of aquatic insect species. Studies with biting flies, mosquitoes and other dipteran vectors have established that dipterans are highly sensitive (Table 6.1).

Table 6.1. Summary of laboratory studies comparing the toxicities of synthetic pyrethroids and selected insecticides to larval dipterans.

Insecticide	Species Tested	Stage	Test Type[a]	24 h LC_{50} $\mu g\ L^{-1}$	Slope of Concentration - Response Curve	Reference
cypermethrin	mosquito, *Culex pipiens quinquefasciatus*	4th instar	S	0.07	1.69	Mulla *et al*. 1978
	mosquito, *Aedes aegypti*	larva	S	0.03	-	Stephenson 1982
	midge, *Chironomus thummi*	"	S	0.1-0.3	-	"
	phantom midge, *Chaoborus flavicans*	"	S	0.03	-	"
deltamethrin	mosquito, *Culex p. quinquefasciatus*	4 instar	S	0.02	1.68	Mulla *et al*. 1980
	mosquito, *Culex tarsalis*	"	S	0.06	1.21	"
	" , *Culiseta incidens*	"	S	0.30	2.37	"
	" , *Aedes nigromaculis*	"	S	0.20	2.19	"
	" , *Aedes taeniorhynchus*	"	S	0.05	1.56	"
	" , *Psorophora columbiae*	"	S	0.10	2.45	"
	" , *Culex pipiens pipiens*	"	S	0.19	2.48	Rettich 1979
	" , *Culex pipiens molestus*	"	S	0.09	2.05	"
	" , *Culiseta annulata*	"	S	0.23	1.84	"
	" , *Aedes cantans*	"	S	0.03	2.70	"
	" , *Aedes sticticus*	"	S	0.02	2.57	"
	" , *Aedes vexans*	"	S	0.09	2.95	"
	midge, *Tanytarsus* spp.	"	S	0.016	2.07	Ali *et al*. 1978
	" , *Procladius* spp.	"	S	0.029	2.79	"
	" , *Chironomus decorus*	"	S	0.23	2.54	"
	" , *Chironomus utahensis*	"	S	0.29	2.16	Ali & Mulla 1978a
	" , *Crictopus* spp.	"	S	0.15	2.16	Ali & Mulla 1980
	" , *Dicrotendipes californicus*	"	S	0.14	3.16	"
	blackfly, *Simulium virgatum*	late instar	CF	0.90	1.80	Mohsen & Mulla 1981

Table 6.1 (cont'd)

Insecticide	Species Tested	Stage	Test Type[a]	24 h LC_{50} $\mu g\ L^{-1}$	Slope of Concentration - Response Curve	Reference
fenvalerate	mosquito, Culex p. quinquefasciatus	4th instar	S	4.70	1.53	Mulla et al. 1980
	mosquito, Culex tarsalis	4th instar	S	4.00	1.55	"
	" , Culiseta incidens	"	S	5.50	1.59	"
	" , Aedes nigromaculis	"	S	2.80	2.65	"
	" , Aedes taeniorhynchus	"	S	0.90	1.81	"
	" , Psorophora columbiae	"	S	28.0	2.01	"
	midge, Tanytarsus spp.	"	S	0.022	1.97	Ali et al. 1978
	" , Procladius spp.	"	S	15.0	2.45	"
	" , Chironomus decorus	"	S	4.20	2.99	"
	" , Chironomus utchensis	"	S	4.20	1.53	Ali & Mulla 1978a
	" , Cricotopus spp.	"	S	80.0	3.78	Ali & Mulla 1980
	" , Dicrotendipes californicus	"	S	44.0	2.80	"
	ragionid fly, Atherix sp.	late instar?	CF	0.60 (72 h LC_{50})	-	Anderson 1982
permethrin	mosquito, Culex p. quinquefasciatus	4th instar	S	1.41	1.58	Mulla et al. 1980
	" , Culex tarsalis	"	S	2.00	1.72	"
	" , Culiseta incidens	"	S	3.00	1.48	"
	" , Aedes nigromaculis	"	S	0.50	1.44	"
	" , Aedes taeniorhynchus	"	S	0.50	2.09	"
	" , Psorophora columbiae	"	S	1.50	1.72	"
	" , Culex p. pipiens	"	S	2.10	1.59	Rettich 1979
	" , Culex p. molestus	"	S	1.40	1.72	"
	" , Culiseta annulatus	"	S	5.90	1.59	"
	" , Aedes cantans	"	S	4.70	2.45	"
	" , Aedes sticticus	"	S	2.70	2.07	"
	" , Aedes vixans	"	S	2.10	2.41	"
	" , Culex p. pipiens	"	S	1.90	2.83	Gaaboub & Abu-Hashish 1981
	" , Aedes aegypti	"	S	1.29	2.31	"
	" , Culex p. pipiens	3rd&4th instar	S	4.00	1.37	Rubio-Moran et al. 1981
	blackfly, Simulium equinum	late instar	CF	0.80[b]	3.33	Muirhead-Thompson 1978a

Table 6.1 (cont'd)

Insecticide	Species Tested	Stage	Test Type^a	24 h LC_{50} µg L^{-1}	Slope of Concentration - Response Curve	Reference
carbaryl	mosquito, *Culex p. pipiens*	4th instar	S	75	1.77	Rawash *et al*. 1975
	" , *Aedes cantans*	"	S	377	1.86	Rettich 1977
	" , *Aedes vexans*	"	S	322	1.63	"
	" , *Aedes excrucians*	"	S	145	3.02	"
	" , *Aedes communis*	"	S	168	2.59	"
	" , *Aedes punctor*	"	S	298	2.10	"
	" , *Culex p. pipiens*	"	S	333	1.55	"
	" , *Culex p. molestus*	"	S	418	1.80	"
	" , *Culiseta annulata*	"	S	180	1.62	"
	" , *Aedes aegypti*	2nd instar	S	1.90 (96 h)	-	Lakota *et al*. 1981
	phantom midge, *Chaoborus* sp.	larvae	S	650	1.90	Bluzat & Seuge 1979
	midge, *Chironomus plumosus*	larvae	S	10 (48 h)	-	Sanders *et al*. 1983
chlorpyrifos	mosquito, *Culex p. quinquefasciatus*	4th instar	S	0.30	-	Ludwig & McNeil 1966
	" , *Aedes aegypti*	"	S	1.002	2.29	"
	" , *Anopheles freeborni*	"	S	3.5-14.0	3.8-6.7	Womeldorf *et al*. 1972
	" , *Aedes cantans*	"	S	1.1	1.58	Rettich 1977
	" , *Aedes vexans*	"	S	1.0	1.79	"
	" , *Aedes excrucians*	"	S	3.3	1.31	"
	" , *Aedes communis*	"	S	1.2	1.54	"
	" , *Aedes punctor*	"	S	2.7	1.66	"
	" , *Aedes sticticus*	"	S	0.5	1.34	"
	" , *Culex p. pipiens*	"	S	1.6	1.54	"
	" , *Culex p. molestus*	"	S	1.2	1.20	"
	" , *Culiseta annulata*	"	S	3.2	1.40	"
	midge, (species not given)	larvae	S	0.50	2.61	Whitney 1968, cf. NRCC 1978a

Table 6.1 (cont'd)

Insecticide	Species Tested	Stage	Test Type[a]	24 h LC_{50} $\mu g\ L^{-1}$	Slope of Concentration - Response Curve	Reference
chlorpyrifos	" , *Tanytarsus* spp.	"	S	0.7	3.16	Ali *et al*. 1978
	" , *Chironomus decorus*	"	S	0.6	1.96	"
	" , *Procladius* spp.	"	S	6.0	2.32	"
	" , *Chironomus utahensis*	"	S	1.2	2.45	Ali & Mulla 1978a
	" , *Cricotopus* spp.	"	S	3.5	2.03	Ali & Mulla 1980
	" , *Dicrotendipes californicus*	"	S	7.0	1.57	"
	blackfly, *Simulium* spp.	late instar	CF	9.7[b]	1.55	Muirhead-Thompson 1970
diazinon	mosquito, *Aedes cantans*	4th instar	S	35.6	1.32	Rettich 1977
	" , *Aedes vexans*	"	S	37.9	1.46	"
	" , *Aedes punctor*	"	S	69.5	1.56	"
	" , *Culex p. pipiens*	"	S	24.3	1.38	"
	" , *Culex p. molestus*	"	S	30.8	1.78	"
	" , *Culiseta annulata*	"	S	62.3	1.51	"
malathion	mosquito, *Culex p. pipiens*	4th instar	S	3.40	1.82	Rawash *et al*. 1975
	" , *Aedes cantans*	"	S	48.8	2.52	Rettich 1977
	" , *Aedes vexans*	"	S	26.1	1.60	"
	" , *Aedes excrucians*	"	S	30.0	1.66	"
	" , *Aedes communis*	"	S	38.0	1.55	"
	" , *Aedes punctor*	"	S	44.0	1.61	"
	" , *Aedes sticticus*	"	S	15.5	1.35	"
	" , *Culex p. pipiens*	"	S	32.0	1.45	"
	" , *Culex p. molestus*	"	S	34.0	1.39	"
	" , *Culiseta annulata*	"	S	24.5	1.46	"
	" , *Culex p. pipiens*	"	S	68.0	1.32	Gaaboub *et al*. 1981
	" , *Culex p. pipiens*	3rd&4th instar	S	42.0	1.37	Rubio-Moran *et al*. 1981

Table 6.1 (cont'd)

Insecticide	Species Tested	Stage	Test Type[a]	24 h LC_{50} $\mu g\ L^{-1}$	Slope of Concentration - Response Curve	Reference
malathion	midge, *Tanytarsus* spp.	4th instar	S	7.4	3.61	Ali *et al*. 1978
	" , *Chironomus decorus*	"	S	7.8	2.05	"
	" , *Procladius* spp.	"	S	100.0	3.31	"
	" , *Chironomus utahensis*	"	S	3.0	3.13	Ali & Mulla 1978
	" , *Cricotopus* spp.	"	S	30.0	2.14	Ali & Mulla 1980
	" , *Dicrotandipes californicus*	"	S	80.0	2.56	"
methomyl	midge, *Chironomus plumosus*	4th instar	S	88.0 (48 h EC_{50})	-	Sanders *et al*. 1983
methoxychlor	mosquito, *Culex p. pipiens*	4th instar	S	8.9	1.56	Rettich 1977
	" , *Culex p. molestus*	"	S	18.9	1.47	"
	" , *Culiseta annulatus*	"	S	38.3	1.36	"
	" , *Aedes punctor*	"	S	63.1	1.56	"
	" , Aedes cantans	"	S	31.5	1.50	"
	midge, *Chironomus tentans*	"	CF	1.62 (96 h TL_{50})	-	Merna & Eisele 1973
	blackfly, *Simulium damnosum*	late instar	S	25.0	1.70	Thompson 1975
trichlorfon	midge, Chironomus plumosus	4th instar	S	0.10 (48 h EC_{50})	-	Sanders *et al*. 1983

[a] S = static; CF = flowthrough

[b] 1 hour exposure with survival monitored at 24 h post-exposure

As larval dipterans, particularly chironomids, are important prey for ducklings, their sensitivity to insecticides provides a potential key to evaluating indirect trophic impacts to waterfowl. Data comparing 24 h LC_{50} values for larvae of the mosquito (Culex pipiens quinquefasciatus) exposed to four synthetic pyrethroids, showed deltamethrin (decamethrin) and cypermethrin to be the most toxic (Mulla et al. 1978) (Table 6.1). The toxicity of permethrin was one order of magnitude lower and fenvalerate was the least toxic. Deltamethrin and cypermethrin were lethal at 24 h static exposure at levels less than 0.3 $\mu g\ L^{-1}$. Midges, with the exception of Tanytarsus and Procladius, were slightly less sensitive to deltamethrin than mosquitoes; a similar trend was noted for fenvalerate but could not be determined for permethrin due to a lack of toxicity test data. Fenvalerate toxicity to midges varied widely from species to species. For example, half the test population of Tanytarsus spp. was killed at 0.02 $\mu g\ L^{-1}$ within 24 h while other midge taxa were 200 to 8,000 times more tolerant.

The toxicity of permethrin to mosquitoes ranged from 0.50-5.90 $\mu g\ L^{-1}$ for 24 h exposures. Values for fenvalerate were within the same range with one exception. The slopes of the concentration-response curves for synthetic pyrethroid insecticides ranged from 1.21 to 3.78, indicating a relatively narrow tolerance distribution within test populations. All other things being equal, as the slope becomes steeper the fiducial limits decrease (see Hodson et al. 1977).

Toxicity test data for dipteran larvae exposed to carbaryl, chlorpyrifos, diazinon, malathion and methoxychlor are presented in subsequent sections of Table 6.1. None of these insecticides are as toxic to dipteran larvae as the pyrethroids except for chlorpyrifos which showed greater or equal toxicity to midges than either permethrin or fenvalerate. Malathion, diazinon and methoxychlor were less toxic (24 h, LC_{50} = 3.0-100 $\mu g\ L^{-1}$) and carbaryl was the least toxic (24 h, LC_{50} generally >100 $\mu g\ L^{-1}$).

In general, the sensitivity of dipteran larvae to acute exposures of synthetic pyrethroids is 10 to 1,000 higher than for most organochlorine, organophosphorus and carbamate insecticides in current use. These comparisons suggest the extreme vulnerability of natural larvae populations of mosquitoes and midges to synthetic pyrethroids, particularly deltamethrin.

The data base for insect orders other than Diptera is not large. A summary of available toxicity data for aquatic taxa, shows that all groups are relatively sensitive to synthetic pyrethroid insecticides (Table 6.2). Mayfly nymphs (Ephemeroptera) were among the most sensitive groups tested (<24 h, LC_{50} <1 $\mu g\ L^{-1}$ for cypermethrin, deltamethrin and permethrin). Adult aquatic bugs and beetles (Hemiptera, Coleoptera) were less sensitive (24 h, LC_{50} <5 $\mu g\ L^{-1}$). Caddisflies (Trichoptera) displayed a range of sensitivities among species for a single insecticide (Muirhead-Thompson 1978a) and a factor of 100 difference between approximate LC_{50} values for

Table 6.2. Summary of toxicity data for synthetic pyrethroid effects on non-target aquatic insects.

Insecticide	Species Tested	Size/Stage	Test Type[a]	Toxic Effect Reported	Value µg L^{-1} (fudicial limits)	Reference
cypermethrin	mayfly, *Cloeon dipterum*	nymph	S	24 h LC_{50}	0.6 (0.3-1)	Stephenson 1982
	beetle, *Gyrinus natator*	adults	S	24 h LC_{50}	5	"
	water boatman, *Corixa punctata*	adults	S	24 h LC_{50}	5	"
deltamethrin	mayfly, *Baetis parvus*	nymph, late instar	S	LC_{50} (24 h after 1 h exposure)	0.4	Mohsen & Mulla 1981
	caddisfly, *Hydropsyche californica*	larva, late instar	S	LC_{50} (24 h after 1 h exposure)	0.4	"
fenvalerate	mayfly, *Ephemerella* sp.	nymph	CF	96 h LC_{50}	0.93	Anderson 1982
	stonefly, *Pteronarcys dorsata*	nymph	CF	72 h EC_{50}, immobilization	0.13 (0.09-0.17)	"
permethrin	burrowing mayfly, *Hexagenia rigida*	35 mm, nymph	S	6 h LC_{50}	0.58-2.06	Friesen *et al*. 1983
	mayfly, *Baetis rhodani*	nymph	CF	LC_{90-95} (24 h after 1 h exposure)	1	Muirhead-Thompson 1978a
	predacious diving beetle, *Erectes sticticus*	larva	S	24 h TLm	5.8	Jeyasingam *et al*. 1978
	caddisfly, *Hydropsyche pellucidula*	larva	CF	LC_{90-95} (24 h after 1 h exposure)	100	Muirhead-Thomson 1978a
	" , *Brachycentrus subnubilis*	larva	CF	LC_{90-95} (24 h after 1 h exposure)	1	"
	" , *Brachycentrus* sp.	larva	CF	21-day LC_{50}	0.2	Anderson 1982

[a] S = static; CF = flowthrough

deltamethrin and permethrin. The response data for stoneflies (Plecoptera), mayflies, and damsel- and dragonflies (Odonata) to non-pyrethroid insecticides is summarized in Table 6.3. Although there is little direct overlap in test species between the pyrethroid, organochlorine, organophosphorous and carbamate compounds tested, the four synthetic pyrethroids appear to be the most toxic of the insecticides reviewed (24 to 96 h, $LC_{50} < 1 \mu g\ L^{-1}$). The only other compound to produce an LC_{50} in the same range was chlorpyrifos which killed half of an exposed stonefly population (Pternarcella baddia) within 96 h.

It is difficult to make broad generalizations based on the available laboratory toxicity data on aquatic insect groups for several reasons. Firstly, very few species have been tested. Secondly, the test species used to evaluate the toxicity of pyrethroid insecticides are not, in most cases, the same as those tested with other insecticides. Thirdly, chosen test species are not always representative of those groups likely to receive maximum exposure or those groups ecologically important to trophic functions as prey for waterfowl and fish. However, the extreme toxicity of the synthetic pyrethroids is evident.

An important observation is the extremely rapid time to response after exposure to pyrethroids, even at very low exposure concentrations. Stephenson (1982) reported that a number of species lost their ability to respond to tactile stimuli with their normal escape response after 2 h exposures to cypermethrin at levels between 0.07-0.5 $\mu g\ L^{-1}$. This impaired response may be, to some extent, reversible. A drift response has been demonstrated for mayflies exposed to as little as 0.5 $\mu g\ L^{-1}$ permethrin for 30 min (Muirhead-Thompson 1978a). There is also one report of a delayed impact to short exposures of permethrin. The burrowing mayfly (Hexagenia rigida), exposed for as little as 6 h in containers with contaminated water and sediment and then removed to clean containers, showed a significant increase in treatment-related mortality from 4 to as long as 8 weeks after exposure (Friesen et al. 1983). The relevance of an extremely rapid and irreversible response pattern to short-term exposure (<24 h), and of delayed mortality not observed within the standard test period, to the interpretation of laboratory toxicity test data and the design of field studies requires further scrutiny (see Section 6.3.5).

6.1.2 Other aquatic invertebrates

A summary of toxicity data for a small number of aquatic crustaceans and molluscs exposed to synthetic pyrethroids is presented in Table 6.4 and more detailed compilations of insecticide tests comparing acute effects on daphnids and amphipods are provided in Tables 6.5 and 6.6, respectively. Synthetic pyrethroids were extremely toxic to all arthropods tested but had minimal effects on the snail, Heliosoma trivolvis. It is again important to note that small crustaceans such as Gammarus, as well as

Table 6.3. Summary of selected toxicity data for aquatic insects exposed to insecticides.

Insecticide[a]	Species Tested		Size/Stage	Test Conditions[b]		Toxicity Reported	Value $\mu g\ L^{-1}$ (95% fiducial limits)	Reference
azinphos methyl	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	25 (21-29)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	8 (6-10)	"
	"	"	nymph	S	-	48 h LC_{50}	8	FWPCA 1968
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	1.5 (0.8-2.7)	Sanders & Cope 1968
	"	*Pteronarcys dorsata*	nymph	S	-	96 h LC_{50}	21.1	Bell 1971
	"	"	nymph	-	-	30 d LC_{50}	4.9	"
	mayfly	*Cloeon dipterum*	6.2 mm	SR	18°C	48 h LC_{50}	12.0 (9.9-14.1)	Dortland 1980
	"	"	6.2 mm	SR	18°C	21 d LC_{50}	3.4	"
	dragonfly	*Ophiogomphus rupinsulensis*	naiad	S	-	96 h LC_{50}	12	Bell 1971 *in* Verschueren 1983
	"	"	naiad	-	-	30 d LC_{50}	2.2	"
carbaryl	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	30 (22-40)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	13 (10-16)	"
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	4.8 (3.0-7.7)	"
	"	*Pteronarcella baddia*	15-20 mm	S	15.5°C	24 h LC_{50}	5.0 (3.6-7.0)	"
	"	"	15-20 mm	S	15.5°C	48 h LC_{50}	3.6 (2.9-4.8)	"
	"	"	15-20 mm	S	15.5°C	96 h LC_{50}	1.7 (1.4-2.4)	"
	"	"	nymph	S	12°C	96 h LC_{50}	13	Woodward & Mauck 1980
	mayfly	*Cloeon sp.*	nymph	S	20°C	48 h LC_{50}	480	Bluzat & Seuge 1979
	"	*Cloeon dipterum*	9.3 mm	S	25°C	48 h TLm	370	Hashimoto & Nishiuchi 1981
	dragonfly	*Orthethrum albistylum speciosum*	23 mm	S	25°C	48 h TLm	430	"

Table 6.3 (cont'd)

Insecticide	Species Tested		Size/Stage	Test Conditions^b		Toxicity Reported	Value $\mu g\ L^{-1}$ (95% fiducial limits)	Reference
chlorpyrifos	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	50 (38-65)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	18 (12-26)	"
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	10 (7-13)	"
	stonefly	*Pteronarcella baddia*	15-20 mm	S	15.5°C	24 h LC_{50}	4.2 (3.0-6.0)	"
	"	"	15-20 mm	S	15.5°C	48 h LC_{50}	1.8 (1.2-2.6)	"
	"	"	15-20 mm	S	15.5°C	96 h LC_{50}	0.38 (0.30-0.49)	"
EC 22.5%	mayfly	*Baetis rhodani*	nymph	CF	17°C 17°C	LC_{90-95} (24 h after 1 h exposure)	10-20	Muirhead-Thompson 1978b
EC 22.5%	dragonfly	*Agrion* sp.	naiad	CF		LC_{90-95} (24 h after 1 h exposure)	200	"
diazinon	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	155 (115-209)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	60 (42-80)	"
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	25 (20-51)	"
	"	*Pteronarcys dorsata*	nymph	-	-	30 d LC_{50}	4.6	Bell 1971 in Verschueren 1983
	mayfly	*Cloeon dipterum*	nymph	S	18.2°C	48 h LC_{50}	7.5	Dortland 1980
	"	*Ephemerella subvaria*	nymph	-	-	30 d LC_{50}	1.05	Bell 1971 in Verschueren 1983
	damselfly	*Lestes congener*	late instar	S	25°C 7.4pH	96 h LC_{50}	50	Federle & Collins 1976
	dragonfly	*Ophiogomphus rupinsulensis*	naiad	-	-	30 d LC_{50}	2.2	Bell 1971 in Verschueren 1983
dimethoate	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	510 (350-740)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	140 (110-170)	"
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	43 (36-51)	"
	dragonfly	*Ophiogomphus rupinsulensis*	nymph	-	-	96 h LC_{50}	12	USEPA 1973

Table 6.3 (cont'd)

Insecticide[a]		Species Tested	Size/Stage	Test Conditions[b]		Toxicity Reported	Value µg L^{-1} (95% fiducial limits)	Reference
malathion	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	35 (23-43)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	20 (15-27)	"
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	10 (7-13)	"
	"	*Pteronarcys dorsata*	nymph	-	-	30 d LC_{50}	11.1	Bell 1971 *in* Verschueren 1983
	"	*Pteronarcella baddia*	15-20 mm	S	15.5°C	24 h LC_{50}	10 (6.7-15)	Sanders & Cope 1968
	"	"	15-20 mm	S	15.5°C	48 h LC_{50}	6 (4.1-8.7)	"
			15-20 mm	S	15.5°C	96 h LC_{50}	1.1 (0.78-1.5)	"
	mayfly	*Baetis* sp.	nymph	S	21°C	48 h LC_{50}	6	Cope 1966
	damselfly	*Lestes congener*	late instar	S	25°C	24 h LC_{50}	300	Federle & Collins 1976
	"	*Lestes* sp.	naiad	S	15°C	96 h LC_{50}	10	Johnson & Finley 1980
	dragonfly	*Ophiogomphus rupinsulensis*	naiad	-	-	30 d LC_{50}	0.52	Bell 1971
methomyl	stonefly	*Pteronarcella* sp.	nymph	S	7° C	96 h LC_{50}	69 (34-143)	Johnson & Finley 1980
	"	*Skwala* sp.	nymph	S	7° C	96 h LC_{50}	34 (27-44)	"
methoxychlor	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	30 (26-36)	Sanders & Cope 1968
	"	"	30-35 mm	S	15.5°C	48 h LC_{50}	8 (5-14)	"
	"	"	30-35 mm	S	15.5°C	96 h LC_{50}	1.4 (1.1-1.8)	"
	"	*Pteronarcella baddia*	nymph	S	10°C	96 h LC_{50}	5.0 (3.9-6.3)	Johnson & Finley 1980
	"	*Stenonema candidum*	nymph	CF	17°C	48 h LC_{50}	1.99-5.66	Bender & Eisele 1971
	mayfly	"	nymph	CF	17°C	96 h LC_{50}	1.83-2.10	"
	"	*Stenonema interpunctatus*	nymph	CF	17°C	96 h TL_{50}	1.96 (1.09-3.11)	Merna & Eisele 1973
	"	*Stenonema terminatum*	nymph	CF	17°C	14 d TL_{50}	0.41	"

Table 6.3 (cont'd)

Insecticide[a]		Species Tested	Size/Stage	Test Conditions[b]		Toxicity Reported	Value µg L^{-1} (95% fiducial limits)	Reference
naled	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	27 (21-35)	Sanders & Cope 1968
		"	30-35 mm	S	15.5°C	48 h LC_{50}	16 (13-19)	"
		"	30-35 mm	S	15.5°C	96 h LC_{50}	8 (5.8-11)	"
oxydemeton-methyl	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	960 (690-1340)	Sanders & Cope 1968
		"	30-35 mm	S	15.5°C	48 h LC_{50}	150 (130-180)	"
		"	30-35 mm	S	15.5°C	96 h LC_{50}	35 (30-39)	"
trichlorfon	stonefly	*Pteronarcys californica*	30-35 mm	S	15.5°C	24 h LC_{50}	320 (200-512)	Sanders & Cope 1968
		"	30-35 mm	S	15.5°C	48 h LC_{50}	180 (128-252)	"
		"	30-35 mm	S	15.5°C	96 h LC_{50}	35 (22-55)	"
		Pteronarcys *baddia*	15-20 mm	S	15.5°C	24 h LC_{50}	50 (39-64)	"
		"	15-20 mm	S	15.5°C	48 h LC_{50}	22 (16-29)	"
		"	15-20 mm	S	15.5°C	96 h LC_{50}	11 (8-16)	"

---

[a] All insecticide compounds tested were technical grade unless formulation indicated

[b] S = static, CF = flow through, SR = static renewed

Table 6.4. Summary of toxicity data for synthetic pyrethroid effects on aquatic crustaceans and molluscs.

Insecticide	Species Tested	Size/Stage	Test Type[a]	Toxic Effect Reported	Value $\mu g\ L^{-1}$ (95% fiducial limits)	Reference
cypermethrin	waterflea, *Daphnia magna*	L 24 h	S	24 h LC_{50}	2 (1-5)	Stephenson 1982
	amphipod, *Gammarus pulex*	3-8 mm	S	24 h LC_{50}	0.1 (0.08-0.2)	"
	isopod, *Asellus* spp.	3-8 mm	S	24 h LC_{50}	0.2 (0.1-0.4)	"
	mite, *Piona carnea*	adults	S	24 h LC_{50}	0.05 (0.03-0.08)	"
	shrimp, *Crangon septomspinosa*	1.3 g	SR	96 h LC_{50}	0.01	McLeese *et al*. 1980
	lobster, *Homarus americanus*	450 g	SR	96 h LC_{50}	0.04	"
deltamethrin	lobster, *Homarus americanus*	450 g	SR	96 h lethal threshold	0.0014	Zitko *et al*. 1979
fenvalerate	amphipod, *Gammarus pseudolimnus*		S	24 h LC_{50}	0.13 (0.10-0.17)	Anderson 1982
	shrimp, *Crangon septemspinosa*	1.3 g	SR	96 h LC_{50}	0.04	McLeese *et al*. 1980
	lobster, *Homarus americanus*	450 g	SR	96 h LC_{50}	0.14	"
	snail, *Heliosoma trivolvis*		S	No mortality, 23 days exposure		Anderson 1982
permethrin	waterflea, *Daphnia magna*	1-2 mm	S	24 h C_{50}	0.2-0.6	Stratton & Corke 1981
	amphipod, *Gammarus pulex*	immature	CF	LC_{90-95} (24 h after 1 h exposure)	1.0	Muirhead-Thompson 1978a
	shrimp, *Crangon septemspinosa*	1.3 g	SR	96 h LC_{50}	0.13	McLeese *et al*. 1980
	lobster, *Homarus americanus*	450 g	SR	96 h LC_{50}	0.73	"
	crayfish, *Procambarus clarkii*	newly hatched	S	96 h LC_{50}	0.39	Jolly *et al*. 1978
	snail, *Heliosoma trivolvis*	0.09-0.3 g	S	96 h, no observed effect conc.	0.33	Spehar *et al*. 1983

[a] S = static; SR = static renewal; CF = flowthrough

aquatic snails like Helisoma, are important components of the diet of laying hens and newly hatched ducklings. Inter-insecticide comparisons are difficult because of the lack of reported toxicity values for all of the pyrethroids to such standard test organisms as Daphnia and Gammarus. The data base for deltamethrin is particularly scant. Daphnids appear to to be less sensitive to the synthetic pyrethroids (24 h, $LC_{50} < 10 \mu g L^{-1}$) than to other insecticides such as azinphos methyl, diazinon and malathion. Gammarids are generally more sensitive than daphnids to all insecticides and are particularly sensitive to the pyrethroids.

The data for crustaceans presents a confusing picture as to the relative toxicities of the various pyrethroids. Toxicity values for Daphnia suggest that permethrin or fenvalerate are the most lethal, while mortality data for Gammarus implicates cypermethrin as the most toxic. The only studies directly comparing the toxicities of various pyrethroids to crustaceans are for the lobster (Homarus americanus) and shrimp (Crangon septemspinosa) (Zitko et al. 1979, McLeese et al. 1980). These data suggest that deltamethrin is the most toxic pyrethroid tested with a 96 h lethal threshold of 0.0014 $\mu g L^{-1}$, a value approaching the levels of analytical detection of the insecticide in water.

A report by Doma & Evered (1977) suggested that the "resting egg" stage (ephippia) of Daphnia may be relatively tolerant to pyrethroids. Permethrin concentrations producing immobilization to first instars at 48 h exposure were as much as 90 times lower than those affecting Daphnia magna ephippia (108 $\mu g L^{-1}$). The additional tolerance of the ephippia stage, adapted to facilitate population survival under conditions of environmental stress (i.e. extreme temperature, drought) may provide a form of protection to the insecticide-exposed cladoceran community.

Observations of crustaceans during acute toxicity tests indicate that immobilization occurs extremely rapidly following pyrethroid exposure but, unlike the effects on some aquatic insects, death does not occur immediately. Abram et al. (1980) reported paralysis of the isopod Asellus aquaticus in less than one hour when exposed to 1 $\mu g L^{-1}$ permethrin. However, few organisms died in less than 15 h even at exposures as high as 1,000 $\mu g L^{-1}$. The authors suggested that relatively short exposure periods may be required for immobilization but that a lag may occur before death even if the ultimately fatal dose is absorbed (but presumably not at the site of toxic action). Mortality may also occur due to increased passive uptake during the lag period. No information is provided on the reversibility of the paralytic process, although field observations confirm recovery of some affected organisms within 24 h (Takken et al. 1978), and laboratory studies by Muir et al. (1985) demonstrated recovery of immobilized chironomids upon removal from water containing, respectively, 0.63, 0.20 and 0.18 ng ml^{-1} cypermethrin, deltamethrin and permethrin. No recovery was observed at initial concentrations 1 to 10 times higher. These speculations have important implications on toxicity testing protocols for the pyrethroid insecticides. Current standard test protocols do not make allowances for short exposure periods (as might occur in nature), delayed mortality or validation of appropriate analytical models of invertebrate toxicity.

Table 6.5. Summary of selected toxicity data for cladocerans exposed to insecticides.

Insecticide[a]	Species Tested	Size/Stage	Test Conditions[b]		Toxicity Reported	Value $\mu g\ L^{-1}$ (95% fiducial limits)	Reference
Synthetic Pyrethroids							
cypermethrin	*Daphnia magna*	24 h	S	18°C	2 h EC_{50} swim when touched	5	Stephenson 1982
	"	24 h	S	18°C	24 h LC_{50}	2 (1.5)	"
deltamethrin	*Daphnia magna*	-	S	-	24 h LC50	8.0	Hoechst Canada, Inc. unpublished
	"	-	S	-	48 h LC50	5.0	"
fenvalerate 30% EC	*Daphnia magna*	adult	S	20°C	24 h LC_{50}	6.6	Day, 1986
permethrin	*Daphnia magna*	1-2 mm	S	20°C	48 h LC_{50}	0.2-0.6	Stratton & Corke 1981
Other Insecticides							
azinphos methyl	*Daphnia magna*	-	S	-	48 h EC_{50} immobilization	2.0	FWPCA 1968
	"	24 h	S	18°C	48 h EC_{50} immobilization	1.6 (1.5-1.7)	Dortland 1980
	"	24 h	SR	18°C	21 d EC_{50} immobilization	0.27	"
	Daphnia pulex	1st instar	S	15°C	48 h EC_{50} immobilization	3.2 (1.8-5.8)	Sanders & Cope 1966
	Simocephalus serrulatus	1st instar	S	15°C	48 h EC_{50} immobilization	4.2 (2.9-6.1)	"

Table 6.5 (cont'd)

Insecticide[a]	Species Tested	Size/Stage	Test Conditions[b]		Toxicity Reported	Value $\mu g\ L^{-1}$ (95% fucicial limits)	Reference
carbaryl	Daphnia magna	5 d	S	20°C	48 h LC_{50} immobilization	7.2	Lakota et al. 1981
	"	1st instar	S	17°C	48 h EC_{50} immobilization	5.6 (2.7-12.0)	Sanders et al. 1983
	Daphnia pulex	-	S	17.5°C	3 h TLm	30	Hashimoto & Nishiuchi 1981
	"	1st instar	S	21°C	48 h EC_{50} immobilization	6.0	Cope 1966
	"	-	S	-	48 h EC_{50} immobilization	6.4 (4.5-8.9)	FWPCA 1968
	Daphnia carinata	2-2.5 mm	S	29-32°C	24 h TLm	100	Santharam et al. 1976
	"	2-2.5 mm	S	29-32°C	48 h TLm	35	"
	Simocephalus serrulatus	1st instar	S	21°C	48 h EC_{50} immobilization	8.0	Cope 1966
	"	1st instar	S	15°C	48 H EC_{50} immobilization	7.6 (6.2-9.3)	Sanders & Cope 1966
carbofuran	Daphnia magna		S	-	24 h LC_{50}	20	Sangha 1972
chlorpyrifos	Daphnia magna	adult	S	18°C	24 h LC_{50}	16	Kenaga et al. 1965
demeton	Daphnia pulex	1st instar	S	15°C	48 h LC_{50}	14 (10.4-18.7)	Johnson & Finley 1980
diazinon	Daphnia magna	24 h	S	18°C	48 h EC_{50} immobilization	1.5 (1.3-1.6)	Dortland 1980
EC 48.2%	"	24 h	S	20°C	48 h EC_{50} immobilization	1.25	Dennis et al. 1979
	"	24 h	SR	18°C	21 d EC_{50} immobilization	0.23	Dortland 1980
	Daphnia pulex	-	S	17.5°C	3 h TLm	80	Hashimoto & Nishiuchi 1981
	"	1st instar	S	21°C	48 h EC_{50} immobilization	0.90 (0.67-1.2)	Sanders & Cope 1966
	Simlocephalus serrulatus	1st instar	S	21°C	48 h EC_{50} immobilization	1.8 (1.4-2.2)	"
	"	1st instar	S	15°C	48 h EC_{50} immobilization	1.4 (1.2-1.6)	Johnson & Finley 1980
dimethoate	Daphnia magna	-	S	-	48 h EC_{50} immobilization	2500	FWPCA 1968
	"	-	S	-	48 h LC_{50}	6400	cf. de Heer 1979
	"	2-26 h	S	19°C	48 h LC_{50}	2500	Frear & Boyd 1967

Table 6.5 (cont'd)

Insecticide^a	Species Tested	Size/Stage	Test Conditions^b		Toxicity Reported	Value µg L^{-1} (95% fucicial limits)	Reference
malathion	*Daphnia magna*	2-26 h	S	19°C	26 h LC_{50}	0.9	Frear & Boyd 1967
	"	1st instar	S	15°C	48 h EC_{50} immobilization	1.0 (0.7-1.4)	Johnson & Finley 1980
	Daphnia pulex	1st instar	S	15.6°C	48 h EC_{50} immobilization	1.8 (1.4-2.4)	Sanders & Cope 1966
	"	-	S	-	48 h EC_{50} immobilization	1.8	FWPCA 1968
	Daphnia carinata	2-2.5 mm	S	29-32°C	24 TLm	200	Santharam *et al*. 1976
	"	2-2.5 mm	S	29-32°C	48 TLm	100	"
	Simocephalus serrulatus	1st instar	S	15.6°C	48 h EC_{50} immobilization	3.5 (2.6-4.8)	Sanders & Cope 1966
methomyl	*Daphnia magna*	1st instar	S	17°C	48 h EC_{50} immobilization	8.8 (4.1-19)	Sanders *et al*. 1983
methoxychlor	*Daphnia magna*	2-26 h	S	19°C	26 h LC_{50}	3.7	Frear & Boyd 1967
	Daphnia pulex	1st instar	S	15.6°C	48 h EC_{50} immobilization	0.78 (0.57-1.1)	Sanders & Cope 1966
	Simocephalus serrulatus	1st instar	S	15.6°C	48 h EC_{50} immobilization	5 (3.8-6.6)	"
naled	*Daphnia pulex*	1st instar	S	15°C	48 h EC_{50} immobilization	0.4 (0.2-0.8)	Johnson & Finley 1980
	Simocephalus sp.	1st instar	S	15°C	48 h EC_{50} immobilization	1.1 (1.0-1.2)	"
phosmet 50% WP	*Daphnia magna*	1st instar	S	21°C	48 h EC_{50} immobilization	11 (9-14)	Johnson & Finley 1980
	"	1st instar	S	21°C	48 h EC_{50} immobilization	5.6 (4.2-8.4)	"
trichlorfon	*Daphnia magna*	1st instar	S	17°C	48 h EC_{50} immobilization	0.12 (0.8-0.22)	Sanders *et al*. 1983
	Daphnia pulex	1st instar	S	15.6°C	48 h EC_{50} immobilization	0.18 (0.3-0.25)	Sanders & Cope 1966
	Simocephalus serrulatus	1st instar	S	15.6°C	48 h EC_{50} immobilization	0.70 (0.56-0.87)	"

[a] All insecticides tested were technical grade unless formulation indicated

[b] S = static; SR = static renewal

Table 6.6. Summary of selected toxicity data for amphipods exposed to insecticides.

Insecticide^a		Species Tested	Size/Stage	Test Conditions^b		Toxicity Reported	Value µg L^{-1} (95% fiducial limits)	Reference
Synthetic Pyrethroids								
cypermethrin		Gammarus pulex	3-8 mm	S	18°C	2 h EC$_{50}$ body flecture in response to touch	0.08	Stephenson 1982
		"	"	S	18°C	24 h LC$_{50}$	0.1 (0.08-0.2)	"
		"		S	15°C	24 h LC$_{50}$	0.06-0.13	Shires 1983
fenvalerate		Gammarus pseudolimnaeus	juv,3 months	S	17°C	24 h LC$_{50}$	0.38	Anderson 1982
		"		S	17°C	48 h LC$_{50}$	0.13 (0.11-0.15)	"
		"		S	17°C	96 h LC$_{50}$	0.05 (0.03-0.06)	"
permethrin	25% EC	Gammarus pulex	late instar	CCF	17.5°	LC$_{90-95}$, 24 h after	1.0	Muirhead-Thompson 1978a
Other Insecticides								
azinphos methyl		Gammarus lacustris	2 months	S	21°C	24 h LC$_{50}$	0.56 (0.39-0.81)	Sanders 1969
		"	"	S	21°C	48 h LC$_{50}$	0.25 (0.19-0.35)	"
		"	-	S	-	48 h LC$_{50}$	0.30	FWPCA 1968
		"	2 months	S	21°C	96 h LC$_{50}$	0.15 (0.11-0.20)	Sanders 1969
		Gammarus fasciatus	-	S	21°C	96 h LC$_{50}$	0.38 (0.24-0.59)	Sanders 1972

Table 6.6 (cont'd)

Insecticide^a	Species Tested	Size/Stage	Test Conditions^b		Toxicity Reported	Value µg L^{-1} (95% fiducial limits)	Reference
carbaryl	*Gammarus lacustris*	2 months	S	21°C	24 h LC$_{50}$	40 (32-49)	Sanders 1969
	"	"	S	21°C	48 h LC$_{50}$	22 (16-30)	"
	"	"	S	21°C	96 h LC$_{50}$	16 (12-19)	"
	Gammarus fasciatus	-	S	21°C	96 h LC$_{50}$	26 (16-39)	Sanders 1972
	Gammarus pseudolimnaeus	-	S	12°C pH7.0	96 h LC$_{50}$	4.1-11.9	Woodward & Mauck 1980
	"	mature	S	17°C	96 h LC$_{50}$	16 (12-19)	Sanders *et al*. 1983
	Gammarus pulex	-	S	20°C	24 h LC$_{50}$	35 (32-39)	Bluzat & Seuge 1979
chlorpyrifos	*Gammarus lacustris*	2 months	S	21°C	24 h LC$_{50}$	0.75 (0.47-1.2)	Sanders 1969
	"	"	S	21°C	48 h LC$_{50}$	0.4 (0.32-0.49)	"
	"	"	S	21°C	96 h LC$_{50}$	0.11 (0.08-0.17)	"
	Gammarus fasciatus	-	S		96 h TL$_{50}$	0.32 (0.12-0.90)	Sanders 1972
EC 22.5%	*Gammarus pulex*	late instar	CF	17.5°C	LC$_{90-95}$, 24 h after a 1 h exposure	50-100	Muirhead-Thompson 1978b
demeton	*Gammarus fasciatus*	mature	S	15°C	96 h LC$_{50}$	78 (42-140)	Johnson & Finley 1980
diazinon	*Gammarus lacustris*	2 months	S	21°C	24 h LC$_{50}$	800 (650-860)	Sanders 1969
	"	"	S	21°C	48 h LC$_{50}$	500 (400-620)	"
	"	"	S	21°C	96 h LC$_{50}$	200 (150-280)	"
	Gammarus fasciatus	mature	S	21°C	96 h LC$_{50}$	0.2 (0.15-0.28)	Johnson & Finley 1980
dimethoate	*Gammarus lacustris*	2 months	S	21°C	24 h LC$_{50}$	900 (670-1200)	Sanders 1969
	"	"	S	21°C	48 h LC$_{50}$	400 (250-630)	"
	"	"	S	21°C	96 h LC$_{50}$	200 (150-270)	"

Table 6.6 (cont'd)

Insecticide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxicity Reported	Value µg L^{-1} (95% fiducial limits)	Reference
malathion	*Gammarus lacustris*	~2 months	S 21°C	24 h LC_{50}	3.8 (3.5-4.1)	Sanders 1969
	"	"	S 21°C	48 h LC_{50}	1.8 (1.3-2.4)	"
	"	"	S 21°C	96 h LC_{50}	1.0 (0.55-1.5)	"
	Gammarus fasciatus	-	S 21°C	96 h TL_{50}	0.76 (0.63-0.92)	Sanders 1972
	"	-	S 21°C	120 h TL_{50}	0.48	"
	Gammarus pseudolimneaus	-	- -	30 d LC_{50}	0.023	Bell 1971
methomyl LC 24%	*Gammarus pseudolimneaus*	mature	S 17°C	96 h LC_{50}	720 (535-940)	Sanders *et al*. 1983
	"	"	S 17°C	96 h LC_{50}	920 (660-1300)	"
methoxychlor	*Gammarus lacustris*	~2 months	S 15.5°C	24 h LC_{50}	2.9 (2.2-3.9)	Sanders 1969
	"	"	S 21°C	48 h LC_{50}	1.3 (0.87-1.9)	"
	"		S 21°C	96 h LC_{50}	0.8 (0.56-1.14)	"
	Gammarus fasciatus		S 21°C	96 h TL_{50}	1.9 (1.2-3.1)	Sanders 1972
	Gammarus pseudolimneaus	mature	CF 12°C	96 h TL_{50}	0.75 (0.47-1.22)	Merna & Eisele 1973
naled	*Gammarus fasciatus*	mature	S 15°C	96 h LC_{50}	18 (16-20)	Johnson & Finley 1980
oxydemeton-methyl	*Gammarus fasciatus*	mature	S 15°C	96 h LC_{50}	1000 (600-1700)	Johnson & Finley 1980
		mature	S 21°C	96 h LC_{50}	1000 (980-1100)	Sanders 1972
phosmet	*Gammarus fasciatus*	-	S 21°C	24 h TL_{50}	5.0 (4.0-6.0)	Sanders 1972
	"	-	S 21°C	96 h TL_{50}	2.0 (1.4-2.8)	"

Table 6.6 (cont'd)

Insecticide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxicity Reported	Value g L^{-1} (95% fiducial limits)	Reference
trichlorfon	Gammarus lacustris	~2 months	S 21°C	24 h LC_{50}	92	Sanders 1969
	"	-	S -	48 h LC_{50}	60	FWPCA 1968
	Gammarus pulex	mature	S 21°C	96 h LC_{50}	40 (26-60)	Johnson & Finley 1980

[a] All insecticides tested were technical grade unless formulation indicated

[b] S = static; CF = flow through

6.1.3 Chronic toxicity

The exceedingly high toxicity of pyrethroid insecticides at low levels of exposure suggests the possibility of chronic effects in treated natural ecosystems. The risk may be particularly high for those organisms associated with macrophyte and sediment habitats where pyrethroids accumulate and persist at low ng g^{-1} levels for periods of a year or more (Rawn et al. 1982, 1983, Solomon et al. 1985). However, the issues of bioavailability and subsequent chronic effects have not been sufficiently evaluated in controlled studies. Rawn et al. (1980) reported that permethrin was residually toxic to the mosquito, *Aedes aegypti*, for a period of 72 h in water samples from a treated outdoor pool. Midge larvae (*Chironomus tentans*) concentrated hydrophobic compounds including isomers of synthetic pyrethroids both when they were suspended in water above contaminated sediments and when they were kept within the sediments (Muir et al. 1983, 1985). Bioconcentration factors (BCF) were calculated by dividing the concentration in the whole animal by the average concentration in water after 24 hours of exposure. Bioconcentration factors of larvae within sediments contaminated with 50 ng g^{-1} of pesticide were: 3067 for hexachlorobiphenyl (HCBP), 372 for methoxychlor, 245 for deltamethrin, 152 for fenvalerate, 113 for cis-permethrin, 67 for trans-permethrin, 25 for terbutryn and 18 for triphenylphosphate (TPP). For most compounds BCFs were higher in animals kept in sand than those kept in clay or silt. At higher concentrations (500 ng g^{-1} in sediments), larvae bioconcentrated pesticides in sediment to a greater degree in sediment than in water. Of the compounds tested for bioavailability to *Chironomus* larvae, only the trans isomers of cypermethrin and permethrin were significantly degraded in water (60% of parent compound remained after 12 h) during the 24 h period required to reach a whole animal steady-state for most compounds and no degradation was observed in the sediments (Muir et al. 1983, 1985). The concentration factor for trans-permethrin was corrected for degradation loss in the above study. During accumulation studies, behavioural effects (reduced undulating behaviour) were noted at 5 ng g^{-1} exposure and some mortality was observed within 24 h at 50 ng g^{-1}. These limited data, together with that of Friesen et al. (1983) previously discussed, are strong circumstantial evidence of the potential bioavailability of pyrethroids from sediment release leading to possible chronic impacts on sediment dwellers.

6.1.4 Summary based on laboratory toxicity data

A review of laboratory studies of deltamethrin, cypermethrin, fenvalerate and permethrin indicate that they are highly toxic to aquatic invertebrates. Molluscs are a possible exception, but information on this group is scant. Key food items for ducks, including dipteran larvae and amphipods, are likely to be severely affected by even low levels of the pyrethroid compounds as well as by azinphos methyl, chlorpyrifos, malathion and methoxychlor; only snails are expected to be reasonably unaffected. The lack of chronic test data and information on bioavailability of these insecticides to sediment and macrophyte-dwelling macroinvertebrates prevents the proper estimation of long-term effects due to low-level exposures. The rapid immobilization response of test organisms to pyrethroids, plus indications of delayed mortality in some species, bring into question standard toxicity testing protocol commonly used to develop preregistration data sets for aquatic species. Furthermore, many of the studies submitted in support of registration do not provide test data on standard aquatic test organisms such as Daphnia and Gammarus or on taxa such as chironomids which are important as prey for ducks.

6.2 Model ecosystem and field studies of insecticide impacts

There are relatively few reported studies describing the ecological impacts of the synthetic pyrethroids and other agricultural insecticides on invertebrate community dynamics in natural or semi-natural lentic ecosystems. The studies include microcosm and enclosure studies, controlled toxicity tests in ponds, sloughs and lakes, and monitoring studies of aquatic systems following insecticide spraying for pest or vector control. The latter group includes wetland, agriculture and forestry examples. Each type of study has a specific utility and certain limitations in defining ecological hazards. In general, the strength of efficacy studies lies in the detection of relative species sensitivity under conditions of natural exposure. Comprehensive impact monitoring can also demonstrate patterns of invertebrate community response under specific ecological conditions. However, due to loading variability inherent in large-scale pesticide application and problems of satisfactorily monitoring residues and ecological response in large field plots, these studies cannot, in general, answer questions regarding concentration-response relationships, overall quantitative community impacts and recovery or secondary impacts only detectable under specific time and experimental conditions. Such questions require specific experimental designs under controlled microcosm and field conditions.

Controlled field tests are difficult to conduct, primarily due to experimental design problems stemming from the inherent variability in natural populations. However, the importance of controlled ecosystem experiments in hypothesis testing must be emphasized. When laboratory data indicate extreme potential risk to aquatic species, field toxicity testing is the only correct approach to detection and quantification of statistically significant population and ecosystem response to and recovery from pesticide exposure.

This review concentrates on controlled dose studies of lentic invertebrate communities responses to insecticides. Other studies will be considered only insofar as they present evidence of the sensitivity of invertebrate taxa or of a specific insecticide-environment interaction influencing community response.

6.2.1 Impacts of pyrethroid insecticides on aquatic ecosystems

A summary of model ecosystem and field toxicity tests of pyrethroid insecticides in aquatic ecosystems is presented in Table 6.7. Demonstrated impact was often substantial and most severe within invertebrate communities. In many cases impact was concentration and species dependent.

6.2.1.1 Cypermethrin

Although there are taxonomic differences in invertebrate response to synthetic pyrethroid insecticides in natural aquatic ecosystems, universal mortality within all taxa is expected at high levels of exposure. A 40% EC formulation of cypermethrin applied at 100 g AI ha^{-1} (approximately three times the recommended rate for cereal crop pests) caused widespread planktonic and benthic invertebrate mortality: 21 of 23 macroinvertebrate taxa present in pre-treatment samples were absent two weeks after application (Crossland 1982). Only the snail Lymnaea peregra and larval representatives of three species of the midge family, Chironomidae, were found in significant numbers following treatment. In the treated pond, zooplankton taxa were reduced from eight to two with the total number of individuals declining to less than 10. Substantial numbers of zooplankters were not found in the treated pond until eight weeks after treatment. Chironomid larvae and adult beetles were found in samples four weeks post-treatment but most other insect species did not reappear for up to 10 to 15 weeks after cypermethrin introduction. The large-scale removal of the zooplankton and benthic insects from grazer communities was credited with contributing to the observed increase in biomass of filamentous algae in the treated pond.

Using stainless steel enclosures in a natural pond, Shires (1983) demonstrated that the amphipod, Gammarus pulex, was very sensitive to cypermethrin under field conditions. The 24 h LC_{50} values in the field (0.06 to 0.13 µg L^{-1}) were very similar to that (0.1 µg L^{-1}) reported by Stephenson (1982) for laboratory tests with the same species. Shires (1983) also found that the field LC_{50} value for rainbow trout (92 g AI ha^{-1}) was similar to laboratory data. Crossland (1982), however, reported no observed effects on wild fish populations (rudd, Scardinius erythrophthalmus, and tench, Tinca tinca) in a pond sprayed at 100 g AI ha^{-1}.

The unreplicated impact studies with cypermethrin discussed above present some general trends in invertebrate community response. The number of individuals and species per sample were depressed to near zero within two weeks after treatment. The numbers of species in the treated pond approached the level in the untreated ponds after 8-10 weeks but the total number of individuals per sample of macroinvertebrates had not recovered by

Table 6.7. Selected reports of model ecosystem and field toxicity tests with synthetic pyrethroid insecticides.

Insecticide	System Studied	Dosage	Impacts	Reference
cypermethrin	natural pond divided into two similar ponds	40% EC at 100 g AI ha^{-1} direct application from backpack sprayer	fish bioaccumulation, no toxicity; reduced richness of invertebrates; repopulation of zooplankton within 11-15 weeks; diurnal D.O. changes	Crossland & Stephenson 1979 Crossland 1982
cypermethrin	pond and stream adjacent to agricultural land	70 g AI ha^{-1} UK pond study; 30-45 g AI ha^{-1} French vineyard study; drift application from tractor mounted sprayer	low levels detectable residues; knock down air breathing aquatic insects; enhanced invertebrate drift	Crossland *et al*. 1982
cypermethrin	enclosures in natural pond	5, 50, 500 g AI ha^{-1} single application on to the water surface from handheld sprayer	residues attained peak concentrations 24 h after application; rainbow trout 3 x more sensitive than carp; LD_{50} = 92g AI ha^{-1} and 300 g AI ha^{-1} for trout and carp respectively	Shires 1983
deltamethrin	natural ponds	Decis formulation 10, 50 g AI ha^{-1} single application from ULV sprayer on a small boat	no fish kill; several insect and crustacean families severely depleted or eliminated	Tooby *et al*. 1981
deltamethrin	replicate artificial ponds	10 g AI ha^{-1} single application injected into water	reduced insect emergence	Rawn *et al*. 1983
fenvalerate	laboratory salt water microcosm	0.01, 0.1, 1.0 g L^{-1} nominal concentrations	for estuarine benthic communities total number of species but not total abundance decreased	Tagatz & Ivey 1981
permethrin	small lakes in forested area	35, 140 g AI ha^{-1} single aerial application	some fish mortality; suppression of zooplankton and bottom fauna over several months	Kingsbury 1976

Table 6.7 (cont'd)

Insecticide	System Studied	Dosage	Impacts	Reference
permethrin	ponds in forested area	17.5 g AI ha^{-1} double aerial application within a 6 day interval	substantial reductions in aquatic invertebrates with second application causing further reductions numbers and increase in lag time prior to recovery	Kingsbury & Kreutzweiser 1979
permethrin	streams in forested area	8.8, 17.5, 35, 70 g AI ha^{-1} generally single aerial application	secondary effects on fish diet, growth, reproductive tissues	Kingsbury & Kreutzweiser 1980 Kreutzweiser & Kingsbury 1982
permethrin	limnocorral enclosures in small lake	0.5, 5.0, 50 g L^{-1} mixed into water	mortality to micro and macrozooplankton; rotifers most tolerant; resilience of zooplankton taxa dependent upon species and exposure	Kaushik *et al*. 1985

the sixteenth week when the study was concluded (Crossland & Stephenson 1979, Crossland 1982). The lack of reported data on variability of invertebrate response restricts further generalizations on the recovery of invertebrate diversity within the cypermethrin treated pond.

6.2.1.2 Deltamethrin

The impacts of deltamethrin on aquatic invertebrate communities have not been well tested under field conditions. Rawn et al. (1983) suggested that the number of insects collected in emergence traps post-treatment (10 g AI ha^{-1}) was reduced from pre-treatment levels, but, due to inadequate pre-treatment sampling and other confounding factors, no definite conclusion regarding the impacts of deltamethrin on insect emergence could be drawn. In a controlled dose study where 10 and 50 g ha^{-1} deltamethrin was applied to two natural ponds, Tooby et al. (1981) reported invertebrate impacts at both treatment levels. However, the authors were unable to make a conclusive statement on concentration-response relations because the invertebrate community in the pond at the lower treatment level was quite different in composition and density than that of both the untreated and higher treatment ponds. A comparison between the untreated pond and that receiving 50 g ha^{-1} deltamethrin showed a substantial post-treatment reduction in individuals of all taxa except Ostracoda (Crustacea), Hydrobiidae and Limnaeidae (Mollusca). No chironomid midges, baetid mayflies or corixid waterbugs were present in samples taken two or six weeks after application in the high dose pond, although these insect families were abundant in pre-treatment samples and were consistently found in collections from the untreated pond. Comparisons of community similarity (Czekanowski's coefficient) between treated and reference pond communities indicated a reduction in taxonomic similarity following treatment: 0.64 to 0.17 at the low treatment and 0.47 to 0.19 at the high treatment. No evidence of recovery in similarity was evident six weeks post-treatment at either application rate.

The wide-scale pest and vector studies with deltamethrin confirm the general susceptibility of aquatic insects and crustaceans, and the relative tolerance of turbellarians, oligochaetes and molluscs. Rettich (1980) found that deltamethrin treatment of 0.2-0.5 $\mu g\ L^{-1}$ eliminated daphnids, trichopterans, hemipterans and the isopod (Asellus aquaticus). Dytiscid beetles were slightly less sensitive but were eliminated at 1 $\mu g\ L^{-1}$; copepods, ostracods, chironomids and springtails (Collembola) generally were only affected at treatment rates of $>5\ \mu g\ L^{-1}$. The oligochaete, Rhynchelnis sp., and turbellarian and mollusc species were not affected by four days of field exposure at 20 $\mu g\ L^{-1}$. The several African studies of deltamethrin efficacy in tsetse fly (Glossina spp.) control confirm insect and crustacean sensitivity (e.g. Takken et al. 1978, Smies et al. 1980, Everts et al. 1983).

6.2.1.3 Fenvalerate

There are no relevant pond studies with fenvalerate. Tagatz and Ivey (1981) demonstrated that 10 µg L^{-1} fenvalerate significantly reduced the total number of species but not the total number of individuals in a field community colonizing saltwater microcosms. The numbers and species of arthropods were significantly less in all insecticide concentrations (0.1, 1.0, 10.0 µg L^{-1}) than in the untreated control system.

6.2.1.4 Permethrin

A greater number of controlled dose studies have been conducted with permethrin. Initial aqueous residues in three ponds in the U.K. treated with 210 g AI ha^{-1} 25% EC permethrin ranged from 30 to 650 µg L^{-1} (Drew et al. 1977). The difference in measurable residue concentrations was apparently due to differences in macrophyte cover and insecticide mixing in the water column of small and large ponds: rapid transfer from the water was observed in shallow ponds with thick emergent plant growth (Edwards *et al*. 1977). Mortality within aquatic arthropod taxa was universal and had approached 100% in the initial week post-treatment. Some recovery was noted for damselflies, waterboatmen (Corixidae) and midges as well as copepod and cladoceran zooplankters at one month post-treatment (Wilkinson & Smith 1977); the mayfly, *Cloeon dipterum*, and the isopod, *Ascellus*, were still completely absent. Hydra, aquatic worms (Oligochaeta and Hirundinea) and molluscs were apparently unaffected by the 210 g AI ha^{-1} permethrin application. Although most other arthropod taxa had recovered to pre-treatment levels after one year, populations of mayflies, aquatic beetles and *Asellus* had not completely recovered (Getty & Wilkinson 1978).

Controlled permethrin additions to limnocorrals in a small Ontario lake have been used to assess zooplankton community response to this insecticide (Smith *et al*. 1980, Kaushik *et al*. 198). Triplicate enclosures were exposed in 1979 and again in 1980 with doses of 5.0 µg L^{-1} and 0.5 µg L^{-1}, respectively. The overall impact of permethrin was to significantly lower zooplankton diversity in treated enclosures. Permethrin was acutely toxic to macrozooplankton causing a three-fold decrease in numbers at all concentrations. Rotifer populations declined significantly (factor of 10) only at 5.0 µg L^{-1}. The resilience of different zooplankton taxa was variable and depended on both the species and the permethrin concentration. For example, recovery of Cladocera started on about day 20 post-exposure in both the 0.5 and 5.0 µg L^{-1} treatments and population densities similar to those in the control corrals (treated only with 0.5 L acetone) were reached in 40-50 days. Complete recovery of copepods was more gradual with densities within the range of control populations approached by day 50. Complete recovery in the high treatment corrals was much slower and was not attained until spring of the following year. The decline in the numbers of rotifers at 5.0 µg L^{-1} was followed by a relatively rapid recovery (~3 weeks). At the lower treatment levels, microzooplankton densities began to expand at about day 10 after application and levels peaked at 4500-5000 individuals L^{-1} at day 22.

Extensive field studies of permethrin effects on aquatic ecosystems in forests provide the most comprehensive sets of field impact data for a pyrethroid insecticide. These studies involved aerial applications to a number of small lakes, ponds and streams (reviewed by Kingsbury for the NRCC, 1986). A single overspray of Thomas Lake, Ontario, at 35 g AI ha^{-1} permethrin produced almost complete mortality of surface caged fish (Cyprinidae) between the third and ninth day. Plankton tow net catches showed that cladoceran populations were reduced by more than 99% and copepods were reduced by two-thirds by day nine of post-treatment. A complete recovery in zooplankton abundance was obvious in samples taken 40 days after treatment. Following application, benthic invertebrate populations were substantially reduced primarily due to the large drop in the numbers of phantom midge larvae (Chaoborus) and the smaller decline in chironomids. A single treatment of Lac Tassel, Quebec, at the application rate of 140 g AI ha^{-1} produced a dramatic and relatively long-lasting reduction of zooplankton. No cladocerans were present one week after exposure and population levels remained low for the next two months. Aquatic insect populations showed a gradual decline with the numbers approaching <3 individuals per Eckman grab by day 30 post-treatment. Amphipods and isopods disappeared from bottom samples soon after treatment and showed no sign of reappearance in samples taken three months after spraying. Phantom midge and amphipod populations remained at very low levels for at least two years beyond treatment at 140 g AI ha^{-1}.

The impact of double applications (6 days apart) of permethrin at 17.5 g ha^{-1} on forest ponds and streams was a further reduction in bottom fauna populations following the second application (Kingsbury & Kreutweiser 1979). The lag time to recovery of invertebrate numbers was extended from one to several months with consecutive applications. In ponds, surface-dwelling insects and phantom midge larvae were the most affected and zooplankton were the least stressed.

At low application rates permethrin had minimal direct toxic effects on fish. In contrast, secondary effects in the form of dietary changes have been documented for both salmonid and other fish species at rates of 8.8-17.5 g ha^{-1} (Kingsbury and Kreutweiser 1979). The major effects were immediate gorging by native fish on distressed aquatic and terrestrial insects with subsequent reduced consumption of aquatic insect species for several weeks to months. Changes in the quantity and quality of prey species consumed can affect fish growth and reproduction. Double applications of 17.5 g ha^{1} permethrin caused temporary reductions in both individual growth rates and population densities of juvenile salmon (Kingsbury & Kreutzweiser 1980). However, higher growth rates in later summer made up the initial treatment weight and length differences.

Kreutzweiser and Kingsbury (1982) suggested that secondary nutritional consequences of permethrin spraying may have repercussions on reproduction as the size of reproductive tissue per unit length of fish was significantly reduced in populations in a 70 g ha^{-1} treated stream. No such effect was suggested by data on brook trout at a 35 g ha^{-1} treatment level.

6.2.2 Impacts of carbamate, organophosphorous and organochlorine insecticides on aquatic ecosystems

Data from model ecosystem and field tests of insecticide impacts on lentic systems indicate that most insecticide compounds have at least minimal short-term adverse effects on invertebrate populations. There is, however, a broad range in the severity and duration of impact and in the types of structural and functional changes induced in exposed pelagic and benthic communities. Less severe impacts are reflected in dominance changes whereas more highly toxic insecticides cause indiscriminate invertebrate mortality accompanied by long lag times to community recovery. A number of the more relevant controlled field toxicity tests for non-pyrethroid insecticides are summarized in Table 6.8.

6.2.2.1 Carbamates

A single aerial application of carbaryl at 840 g AI ha^{-1}, a rate equivalent to that used for spruce budworm control but less than that recommended for cereal pests, caused substantial and persistent reductions in amphipod and sensitive insect populations (Hunter *et* al. 1984). Amphipod, mayfly and caddisfly species responded rapidly to carbaryl exposure: many individuals died in the initial 1-4 hours post-spray. Although the amphipod species *Hyallela azteca* and *Crangonyx richmondensis* had failed to recolonize treated ponds 30 months after treatment, the impact on mayfly and caddisfly abundance did not persist into the following year. In addition, the numbers of emerging caddisflies of the species *Phytocentropus hansoni* were reduced within the post-treatment season. Dragonfly and damselfly populations were lower in treated ponds than in controls throughout the two years following carbaryl treatment, possibly due to a combination of direct toxicity and reductions in available prey for naiads. The numbers of chironomid midges, the most abundant taxa in the experimental ponds, were significantly reduced in one of the treated ponds shortly after spray, but no differences were measured in adult emergence.

In conjunction with the study discussed above, Hunter and coworkers examined the impacts of the depression in invertebrate abundance on duckling growth and behaviour. Ducklings on sprayed ponds spent more time searching for food and less time resting than did controls, and decreased growth rates were recorded for Black Duck (*Anas rubripes*) and Mallard (*Anas platyrhynchos*) ducklings on treated ponds. There were no significant treatment effects on duckling survival, however, the ducklings were not on the treated pond during at least half of their critical two-week phase of dependency on aquatic invertebrates as food. This study provides a critical link in the risk analysis discussed later in Section 6.3.

Table 6.8. Selected reports of model ecosystem and field toxicity tests with carbamate, organochlorine and organophosphorous insecticides.

Insecticide	System Studied	Dosage	Impacts	Reference
azinphos methyl	outdoor model ecosystems with sediment and macrophytes	1 mg L^{-1} concentration in water; redosed to this level twice weekly	factors of 100 and 10 reductions in abundances of *Daphnia*, *Simocephalus vetulus*, respectively; recovery in cladoceran numbers was not evident until more than one month after discontinuing treatment	Dortland 1980
carbaryl	woodland ponds; four treated, three untreated controls; some sphagnum ponds with organic substrate and some beaver ponds with mineral substrate	840 g AI ha^{-1} single aerial application	amphipod population reduced from >1000 m^{-2} to <1 m^2; no substantial recovery in population of *Hyabella azteca* through the three-year study period; mayfly populations reduced by 75% immediately after application and caddisfly emergence decreased by two-thirds; dragon- and damselfly larva low numbers through two years possibly due to shortage of prey; most abundant group, midges, not significantly changed; decreased growth rate of ducklings held on treated ponds	Gibbs *et al*. 1984 Hunter *et al*. 1984
carbaryl	woodland streams	single aerial application at 840 g AI ha^{-1} first year followed by 1120 g AI ha^{-1} second year	initial response 170-times increase in drift, benthos samples showed significant decline in stoneflies, mayflies and caddisflies; further reduction in stoneflies second year; true flies (Diptera) and oligochaete worms generally unaffected	Courtemanch & Gibbs 1980

Table 6.8 (cont'd)

Insecticide	System Studied	Dosage	Impacts	Reference
chlorpyrifos	replicate artificial ponds; duplicate of control and four treatments	4.54 g AI ha^{-1}, 22.7 g AI ha^{-1}, 45.4 g AI ha^{-1}, 454 g AI ha^{-1}; three applications at each treatment level at two-week intervals using a handheld sprayer	the cladoceran *Moina micrura* and the copepod *Cyclops vernalis* experienced 100% and 26% mortality, respectivey, after the third treatment at 4.54 g ha^{-1}; no recovery within 30 days after the first application at two highest treatment levels; populations of the rotifer *Asplanchna brightwelli* increased after second and third application at two lowest treatment levels; corixids, the major insect group, were reduced at treatment of 22.7 g ha^{-1} and greater with no recovery obvious at the highest treatment level	Hurlbert *et al*. 1970
chlorpyrifos	replicate artificial ponds; four each at control and two treatment levels	28 g AI ha^{-1} and 280 g AI ha^{-1}; three applications at each treatment level at two-week intervals using a handheld sprayer	crustacean zooplankters *Cyclops* and *Moina* experienced high mortality due to treatment; recovery at low dose required 1-3 weeks and at the high dose 3-6 after final treatment; predaceous insects showed greater reductions than herbivorous forms and recovered more slowly with only 9% and 45%, respectively, of control numbers five weeks after treatment; phytoplankton blooms after treatment possibly due to reduced grazing	Hurlbert *et al*. 1972

Table 6.8 (cont'd)

Insecticide	System Studied	Dosage	Impacts	Reference
chlorpyrifos	artificial ponds; duplicates of controls and two levels of treatment	11.35 g AI ha^{-1}, 56.75 g AI ha^{-1}; two applications at each treatment level approximately one month apart using a handheld sprayer	mean number of insects colonizing plate samplers reduced by 50% at low treatment and 75% at high treatment over 63 days of observation; mayflies and caddisflies most affected and midges least affected, however, midge emergence at 56 g ha^{-1} reduced to less than 30% of that in untreated ponds	Macek et al. 1972
chlorpyrifos	one natural and two artificial ponds	10 μg L^{-1} AI; single application to surface with gentle mixing	numbers of cladoceran reduced to near zero within 24 hours and no evidence of recovery up to 40 days; copepod nauplii density reduced and held near zero for 20 days; algal bloom beginning about 30 days after treatment	Hughes et al. 1980
diazinon	three outdoor experimental channels; control, low and high treatment	continuously dosed at 0.3 and 3 μg L^{-1} for 12 weeks then increased to 6 and 12 μg L^{-1} for four weeks; high treatment was then increased to 30 μg L^{-1} (intended concentrations not reached in latter part of study (>5 g L^{-1}) due to bacterial degradation of diazinon)	no consistent trends in total macroinvertebrate abundance or species diversity with treatments; changes were reported in community structure which were more pronounced with increased concentrations of insecticide; mayfly and damselfly emergence affected at high treatment level	Arthur et al. 1983

Table 6.8 (cont'd)

Insecticide	System Studied	Dosage	Impacts	Reference
malathion	artificial ponds; four replicates, control and two levels of treatment	2, 20 g L^{-1}; four applications mixed into ponds at approximately three-	significant differences among the number of insects colonizing plate samplers; midgeswere most tolerant but were inhibited with low-level multiple exposures	Kennedy & Walsh 1970
methyl parathion	divided natural ponds; three treated and three	100 g L^{-1} AI single application	*Daphnia* populations virtually eliminated (<1 per sample), cyclopods reduced to a lesser extent and diaptomids not significantly reduced from controls; numbers of dipterans and ephemeropterans significantly reduced; there was recovery of populations of midges, mayflies and *Daphnia* by day 90; depressed growth rate of rainbow trout probably due to reductions in insect prey	Crossland & Elgar 1983
methyl parathion	replicate enclosures in pond	2.5 mg L^{-1} single application into the water column	Cladocerans and copepods decreased in abundance; nauplii decreased to near zero; arthropod macroinvertebrates depleted	Stephenson & Kane 1984
methoxychlor	artificial ponds; duplicates of controls and two levels of treatment	10, 40 g L^{-1} single application mixed into ponds	number of taxa sucessfully colonizing plate samplers was lower in treatment ponds (11 in treated and 18 in untreated); total number of organisms increased with treatment; midges were most tolerant and mayflies most severely affected	Kennedy *et al*. 1970

Consecutive annual applications of carbaryl at 840 g AI ha^{-1} the first year and 1120 g AI ha^{-1} the following year to woodland streams caused an initial post-spray increase in invertebrate drift which was 170-times normal levels (Courtemanch & Gibbs 1980). Declines in benthic insect abundance were further aggravated by the second year of treatment. Haines (1981) reported that two applications of 567 g AI ha^{-1} carbaryl spaced seven days apart affected fish diet. The amount of insect food consumed by fish in the treated stream did not change after the first application but increased immediately after the second. Longer-term impacts of low benthic insect abundance were not evident in fish condition factors through 30 days of post-spray observation.

The effects of carbofuran treatments at 25 µg L^{-1} the first year and 50 µg L^{-1} the second year on the zooplankton community of a natural pond were summarized as negligible although no data were presented (Klaasen & Kadoum 1979).

6.2.2.2 Organophosphates

Dortland (1980) reported that azinphos methyl maintained at 1 mg L^{-1} in outdoor tanks caused two- to three-fold reductions in cladoceran abundance, and recovery in population numbers was not evident for more than one month after the termination of treatments. The exposure of invertebrate communities in similar systems to 1 mg L^{-1} parathion caused even more extensive mortality: near total elimination of cladocerans and substantial reductions in the dominant insect, the phantom midge, Chaoborus crystallinus. Neither insecticide significantly affected the numbers of cyclopoid copepods or ostracods. In experimental pond studies of the use of Guthion (azinphos methyl) at 1 mg L^{-1} nominal concentration as a fish eradicator, no pronounced changes were reported for zooplankton abundance but the numbers of larval insects were reduced for the initial 10 days post-exposure (Meyer 1965).

Perhaps the most comprehensive sets of data for field testing of insecticide impacts are for chlorpyrifos. Studies by Hurlbert et al. (1970, 1972) examined insecticide persistence and impacts on crustacean zooplankton and insect species. Populations of the cladoceran Moina micrura and the copepod Cyclops verrulis were severely impacted with 0% and 74% survival, respectively, after three treatments at the low rate of 11.2 g ha^{-1}. No significant recovery in these taxa was obvious within 30 days after the cessation of spraying between 112 g ha^{-1} and 1121 g ha^{-1} (Hurlbert et al. 1970). In a second set of experiments, the relationship between treatment level and recovery for cladocerans, and copepods was described as 1-3 weeks lag for triple applications at 28 g AI ha^{-1} and 3-6 weeks lag at 280 g AI ha^{-1} (Hurlbert et al. 1972). In contrast, rotifer populations increased in treatment ponds after the second and third application of chlorpyrifos. In treated ponds, predaceous insects showed greater reductions than herbivorous species and populations recovered more slowly with 9% and 45%, respectively, of control levels measured five weeks after the conclusion of treatments. Algal blooms were common in treated ponds after the multiple treatment regime.

Studies by Hughes et al. (1980) in natural and artificial ponds confirmed earlier observations of the relative sensitivity of zooplankters and the link of chlorpyrifos to algal blooms. A single application of 10μg L^{-1} reduced the numbers of cladocerans to near zero within 24 hours and delayed recovery for a period of 40 days. Copepod nauplii were reduced by the treatment and held at low levels for approximately 20 days. Hughes et al. (op. cit.) suggested that the algal blooms found in treated ponds beginning at about day 30 occured primarily in response to reduced grazing pressure. The mean number of benthic insects colonizing plate samplers in artificial ponds over 63 days of observation was 50% less than control sample levels with applications of 11.35 g AI ha^{-1} and 75% less at a higher (5x) treatment level (Macek et al. 1972). Mayflies and caddisflies were the most sensitive taxa and midges were the most tolerant. At the application rate of 56.7 g AI ha^{-1}, a level 10 times less than that recommended for chlorpyrifos control of grain pests, emergence of adult midges was reduced to 30% of levels on the untreated pond.

Artificial pond studies by Rawn et al. (1978) indicate that sediment composition may dictate the bioactivity of chlorpyrifos to free swimming aquatic invertebrates. Sandy sediment prolonged the period of 100% control of the mosquito, Culex spp., by a factor of two over that measured in sod and clay-lined ponds, implicating the importance of insecticide adsorption to organic matter and clays in reducing bioavailability to pelagic species. A number of reports of wide-scale chlorpyrifos applications in aquatic pest and vector control confirm severe larvicidal impacts to pest and non-target species (e.g. Ludwig et al. 1968, Washino et al. 1972, Mulla et al. 1973).

The impacts of diazinon on macroinvertebrates in outdoor experimental channels were reported by Arthur et al. (1983) who applied a continuous dosing scheme of 0.3 and 3 μg L^{-1} for 12 weeks, then increased nominal concentrations to 6 and 12 μg L^{-1} for the next four weeks, after which they ceased the low treatment but further increased the high level to 30 μg L^{-1}. The authors found no consistent trends in total abundance or species diversity correlated with treatments. Changes in community composition were more pronounced with increasing concentration in the exposure regime. Mayflies, damselflies, caddisflies and amphipods were listed as sensitive; flatworms, snails, isopods and midges were tolerant. The total number of individuals and species emerging from test channels was not very different but fewer mayflies and damselflies emerged from treated channels. Interpretations were confounded by the inability of the dosing regime to maintain nominal concentrations at the high treatment level. Apparently, repeated low level exposures were selective for microbial populations contributing to the rapid degradation of diazinon. A vector control study with diazinon applied at a rate producing a nominal concentration of 2.4 mg L^{-1} in rice fields provided only selective short-term control of mosquitos by eliminating a Culex but not an Anopheles species (Ree et al. 1981). These limited data suggest that a relatively low risk of substantial impact might be expected from applications of diazinon at recommended pest control rates to pond ecosystems.

Malathion and parathion have more substantive effects on natural aquatic invertebrate communities. Tests of malathion toxicity in replicate artificial ponds receiving multiple applications at three-week intervals at treatment levels of 2 and 20 $\mu g\ L^{-1}$ demonstrated significant reductions in the numbers of insects colonizing plate samplers. Chironomidae, the dominant insect family, were unaffected by a single treatment but showed a significant reduction in both high- and low-treated ponds after the third application of malathion. Heptageniid mayflies formed about 5% of the benthic community in the untreated pond samples but were scarce in treated ponds and failed to recolonize plates after the third application (Kennedy & Walsh 1970).

Aerial application of 150 g AI ha^{-1} malathion to salt marshes resulted in a 50% reduction in aquatic invertebrates, although no significant change in the diversity of organisms was detected (Wan & Wison 1976).

Field studies with parathion indicate that its impact in small lentic ecosystems may be more severe and long-lived than that of malathion. Using natural ponds divided by a concrete wall, Crossland and Elgar (1983) found that a single treatment of methyl parathion at 100 $\mu g\ L^{-1}$ virtually eliminated daphnids from the zooplankton community and reduced the numbers of cyclopoid copepods to a lesser extent. Diaptomids were not significantly affected. The number of free-swimming dipterans and ephemeropterans were reduced as was the dominant benthic taxon, Chironomidae. Populations of Daphnia, midges and mayflies did not fully recover from treatment effects until about day 90 post-treatment.

Methyl parathion applied to a natural lake system at a rate of 3.3 $\mu g\ L^{-1}$ for three treatments at 20-day intervals caused temporary suppression of zooplankton populations (Apperson et al. 1976). The initial two treatments had little effect on copepod abundance. Diaptomus declined by 32% following the second treatment and a concomitant decrease in length suggested that mortality was widespread among larger individuals. Nauplii populations declined in the initial treatment in June and remained below pre-treatment levels for approximately six weeks. All copepods declined after the third application. Daphnia appeared highly susceptible to methyl parathion: numbers declined by 52%, 27% and 20%, within 24 hours after the first, second and third treatments, respectively. The early instars of daphnids were most affected.

Stephenson and Kane (1984) used replicate enclosures in a shallow pond to test the susceptibility of macroinvertebrates to methyl parathion and to examine secondary effects on community structure and function. Of the 14 families of arthropods present in the uncontaminated enclosures, only two were present in samples from the treated enclosures. There was no significant difference in the evenness component of macroinvertebrate diversity. Increases in Chl a fluorescence was correlated with decreases in zooplankton abundance.

6.2.2.3 Organochlorines

Pond studies conducted by Kennedy et al. (1970) with methoxychlor suggested that its impacts on the invertebrate community were less severe than those of malathion (see Kennedy & Walsh 1970). A single application of 10 or 40 µg L^{-1} methoxychlor mixed individually into artificial ponds lowered the numbers of taxa successfully colonizing plate samplers, with 11 groups in treated ponds and 18 groups in untreated ponds, but did not reduce abundance. In contrast to the trend of reduced abundance previously reported, the total number of organisms increased with methoxychlor treatments. More tolerant midge species expanded in abundance at the expense of the more severely impacted mayflies.

In general, field studies of methoxychlor impacts have been concentrated in stream and river systems and have been largely associated with the problem of blackfly control. These studies suggest that mayfly, stonefly and free-living caddisfly immatures are highly sensitive to methoxychlor. However, due to problems associated with insecticide dilution and invertebrate drift (as related to community recovery), the persistence of structural changes in streams is difficult to determine and data is not transferable to lentic ecosystem impact evaluations (see NRCC 1983 for discussion of impact study design in lontic environments).

6.2.3 Critique of field studies

Data collected from controlled tests of insecticide toxicity in the field suggest that substantial impacts on aquatic ecosystems should be expected from applications, at rates recommended for control of cereal crop pests (see Table 6.12), for the synthetic pyrethroid compounds, azinphos methyl, chlorpyrifos, malathion and, to a lesser extent, carbaryl and methoxychlor. The limited diazinon data suggest that it is of low risk. No definitive judgements can be made for the other major use insecticides listed in Table 1.1 due to a lack of field impact data.

The vast differences in test conditions, test ecosystems and sampling techniques make detailed comparisons of the severity and duration of impacts impossible. Certain generalizations can, however, be made. When applied at crop protection rates, there will be a range in the severity of impact of different insecticides on aquatic invertebrates; the effect depends on both the compound and the ecological system. Multiple applications closely spaced can adversely affect even tolerant populations and dampen the resilience of all exposed populations. Recovery is influenced by the availability of colonizers as well as the life history of the species. Thus, the impacts on aquatic invertebrates are more severe and long-lasting when most or all of the habitat is treated as compared to partial treatment of a lake or pond where patches of potential colonizers remain unaffected (see Ali & Mulla 1978b). Data also indicate that the depression in invertebrate abundance can have indirect trophic impacts on fish (Kingsbury & Kreutzweiser 1980, Kreutzweiser & Kingsbury 1982) and some waterfowl species (Hunter et al. 1984).

It is obvious from the available data that the bulk of aquatic impact studies have not adequately addressed sampling and statistical criteria essential to quantifying an appropriately sensitive level of community change. Even some of the controlled toxicity tests cited in Tables 6.7 and 6.8 provide only coarse indications of the acute impact and its duration. Questions of organism life history, habitat differences, sampling bias, the precision of sampling techniques, and the identity of critical ecological links, are often not adequately evaluated prior to selecting an appropriate experimental design. The critical question of primary and secondary biological impacts should receive experimental priority, i.e. find out if there are adverse effects under "worst case" exposure conditions when ecosystem contamination is probable, and then, if necessary, assess the complex problem of exposure under varying application and environmental conditions.

6.3 Ecological Hazard Evaluation

6.3.1 Need for a framework to evaluate hazards from pesticides

The objective of preregistration testing of pesticides must be to gain the information on their environmental behaviour, accumulation patterns and toxicity necessary to predict where a compound is likely to be and for how long (exposure) and to suggest which taxa, communities and ecological processes are most likely affected (toxicity). Such information facilitates analyses providing a relative hazard ranking and subsequent risk estimates for critical components.

Most evaluations of pesticide hazard to wildlife are based primarily on extrapolations of laboratory evidence of direct toxic effects of specific compounds on standard test species to natural populations. Indirect impacts of mediated ecological interactions affecting food and habitat resources have not been a prime consideration. The evaluative process has been largely unstructured and heavily dependent on the ability, based on knowledge, experience and intuition, of the evaluator to draw together exposure and effects data into a qualitative estimate of potential hazard. Such an approach neither clearly articulates to the pesticide producers the data required by the evaluator in the decision-making process nor facilitates consistency or accuracy in hazard predictions. An effective evaluation scheme must involve a co-ordinated analysis of both environmental fate and laboratory and field toxicity data and a decision-making process based on acceptable criteria, as well as the experience of the evaluator. The objective is an evaluation framework with efficient tools that provide consistent and accurate predictions of hazard. A suitable framework should have general utility in predicting direct impacts and should facilitate an analysis of indirect impacts and specific exposure situations with proper modification.

This section contains an analysis of hazard evaluation frameworks and proposes models assessing primary impacts on aquatic fauna. Modification and application of the general models to the assessment of indirect insecticide effects on ducks are discussed as an example of an application which meets current Canadian Wildlife Service management needs.

Evaluation frameworks involving computer modelling techniques are emerging in the fields of environmental hazard/risk analysis. Within these approaches, simulated environmental concentrations are compared to concentrations producing toxic response in laboratory tests. These models are nothing more than a systematic approach to extrapolation which permits explicit consideration of all the assumptions that go into the extrapolation. In the initial stages of development, these models of ecological hazard can only be utilized as a screening tool. It is important to realize that such hazard models must be a compromise of realism (complexity), to provide accuracy of prediction, and simplicity, to provide ease of use and generality of results. For example, at the screening level these models should require a minimum of essential input data. In addition, they must be considered as qualitative predictors of relative hazard, dependent upon the development of benchmark data substantiated by field studies to provide boundaries for rankings of relative risk and an estimate of the extent of hazard. A preliminary hazard evaluation should provide a basis for further calculation of the probability of risk for important taxa or taxonomic groups. Ultimately, a hazard evaluation framework will be judged more acceptable than current procedures if predictions based on its output are more consistent and accurate than current forecasts.

6.3.2 Hazard and risk modelling: objectives and evaluation criteria

For the purpose of evaluating pesticide impacts, hazard modelling involves identifying the segment of the environment and taxa or ecological process most likely to be affected by pest control products. The magnitude and persistence of expected pesticide concentrations in the environment are compared with those concentrations toxic to representative taxa to rank the relative risk of adverse impact (see Kenaga 1982a). Ecological risk analysis is defined as the process of identifying and quantifying the probabilities of adverse changes in a population, community or ecosystem resulting from the use of pesticides. The objectives of risk analysis are thus both identification of problem situations, including relevant interactions, and the prediction of risk in probability terms.

Hazard assessment models have taken two forms: 1) subjective scoring systems quantifying the consensus of expert evaluators (e.g. scoring, Welch & Ross 1982; analytical hierarchy method, Barnthouse et al. 1982) and 2) applied tools for the evaluation employing exposure simulation and extrapolation of toxicity benchmarks (e.g. relative hazard evaluation, Zitko & McLeese 1980; chemical migration and risk assessment (CMRA), Parkhurst et al. 1981; analysis of error extrapolation, Suter et al. 1983). Of particular interest are those techniques that fall within the latter category. They provide the evaluator with a conceptual framework as well as modelling and mathematical tools for estimating the level of hazard and for analyzing some of the sources of uncertainty inherent in these estimates. The analytical tools take the form of computer models to simulate exposure in various receiving systems (e.g. PERSISTENCE, NRCC 1981; EXAMS, Burns et al. 1982; SERATRA, Onishi & Wise 1982; FUGACITY, MacKay & Paterson 1982;

TOXIWASP, Ambrose et al. 1983) and mathematical and statistical techniques to facilitate comparisons of environmental concentrations and toxicological benchmarks to categorize the severity of potential damage. Complex system models incorporating ecological and toxicological interactions are also in the initial stages of development and evaluation (e.g. O'Neill et al. 1982, 1983).

Although hazard modelling appears promising, there is no one accepted model and the few proposed models have not been fully tested. Therefore, the selection of an appropriate approach to the problem of screening insecticides for primary and secondary impacts in aquatic ecosystems was initiated by 1) first defining criteria upon which to judge hazard analysis models, 2) evaluating proposed models with these criteria and 3) modifying current techniques and/or developing techniques to better meet evaluative criteria as well as the specific demands of the waterfowl-insecticides project.

Criteria for evaluating ecological hazard models are contained in the following questions:

1) Does the model have an acceptable theoretical base?

2) Does the approach used clearly define quantifiable and ecologically (and socially) relevant endpoints?

3) Is the model functionally simple and easy to use?

4) Are the data requirements realistic?

5) Can the model be used under a variety of toxicological and environmental conditions, i.e. does it have generality?

Table 6.9 presents a summary of our evaluation of four of the models proposed for hazard analysis. A simple technique such as the quotient method is easy to use and has generality. It involves a direct comparison of estimated or empirically measured environmental concentrations with standard toxicological benchmarks (LC_{50}, MATC). The quotient index is merely the ratio of the estimated ambient concentration to the acute or chronic toxic concentration. The method has no defined theoretical base for choice of either the environmental or toxicological concentration. Interpretation of hazard requires qualitative judgement in classification of quotients. A relative ranking is produced; no estimation of error is included (see Barnthouse et al. 1982 for further evaluation).

The relative hazard index used by Zitko and McLeese (1980) to evaluate the relative risks of forest insecticides to aquatic systems is based on a 6-compartment equilibrium model and a 3-compartment kinetic model to estimate exposure. Relative hazard is calculated by dividing the expected environmental concentration by the 96 h LC_{50} and multiplying this lethality index by the bioconcentration factor. This model has a theoretical base for the estimation of exposure but the theory is more tenuous for the connection of toxicity, bioconcentration and exposure estimates. The relative hazard index is used to rank increasing hazard; no estimates of uncertainty are included.

Table 6.9. Summary evaluation of proposed models of ecological hazard in aquatic ecosystems ("+" = meets criteria; "-" = fails to meet criteria).

Criteria	Models			
	Quotient Method[a]	Relative Hazard Index[b]	Chemical Migration and Risk Assessment[a]	Analysis of Error Extrapolation[a]
theoretical base	-	for persistence calculation	+	for toxicological extrapolations
ease of use	+	+	-	+
data requirements	+	+	-	-
endpoints	lethality to fish	lethality and bioconcentration	lethality to fish	lethality to fish
generality	+	not with toxicity model	only flowing water systems	no consideration of exposure model

[a] See description in Barnthouse et al. 1982

[b] See description in Zitko 1980; Zitko & McLeese 1980

[c] See description in Olsen & Wise 1982; Onishi & Wise 1982; Onishi et al. 1982

With the analysis of extrapolation error method, expected effects on organisms in the field are estimated with appropriate error bounds from observed effects on laboratory test organisms. Major sources of error are taxonomic differences, life-stage differences and differences in the physiological state of organisms. Its potential advantages include the definition of sources of error, consideration of variation in species and life-stage and correction of some biases of toxicological data. No simulation of expected environment concentration is included. Results take the form of estimates of probability of undesired endpoint events but cannot be construed as a prediction of the frequency of those events.

The Chemical Migration and Risk Assessment Model (CMRA) from Battelle Laboratories predicts the frequency of adverse events using frequency analysis to provide statistical summation of time-varying contaminant concentrations and to link simulated aquatic toxicant concentrations and laboratory LC_{50}'s and maximum acceptable toxicant concentrations (MATC) (Olsen & Wise 1982, Onishi & Wise 1982, Onishi et al. 1982). It is the most complex of the models considered and, as such, demands the most input data. Error analysis is included in the exposure model.

One obvious weakness in most of the risk models is the lack of a sound theoretical base for the comparison of exposure and toxicity data and the nearly universal lack of uncertainty estimates. The CMRA model, although perhaps the most sophisticated, is unwieldy for the purpose of screening. None of these models have been thoroughly checked for a systematic and reasonable level of convergence between model predictions and field data.

To overcome some of these limitations, we have developed two hazard assessment models, in conjunction with previous National Research Council work (NRCC 1981), for ease of use and flexibility in handling physico-chemical, degradation and toxicity data. Exposure-toxicity comparisons are based on theory, and uncertainty estimates, although not currently included, are considered for future refinements. Our models have general applicability in ranking the relative hazard of direct adverse impacts of pesticides on aquatic populations and communities. They are primary response models and can be easily modified to predict secondary indirect impacts of pesticides. Model predictions are biased to screen relative risk with a minimum of data under "worst case" conditions of maximum compound bioavailability and toxicity. Additional fate and toxicity data (or improvements in data quality) encourage refinements in hazard prediction and, generally, a lowering of relative risk ranking.

These primary hazard models consider the most likely cause of contamination of an aquatic system from a single application of an insecticide. Two distinctly different phases of bioavailability of applied compounds are expected in water under such conditions. During the initial phase of dissipation, concentrations are high and losses rapid. This is followed by a phase where distribution between environmental compartments has equilibrated, concentrations are low and loss rates are slow. The two persistence phases are depicted in Figure 6.1. The initial phase is descriptive of an acute model of exposure and the equilibrated distribution phase of a chronic exposure model. Likewise, there are two possible models of toxic responses: one based on maximum concentration and complete reversibility of mechanisms below the lethal response threshold, and the other based on integrated exposure and chronic response with accumulation of concentration or damage producing mortality. These divergent fate and toxicity considerations need to be integrated with proper theoretical support to produce acceptable hazard models accounting for the various extremes.

6.3.3 Calculation models for the analysis of ecological hazard

Two indices of relative hazard were derived to assess potential impacts of insecticides on aquatic fauna under acute and chronic exposure conditions in lentic ecosystems. The relative hazard index (RHI) is a dimensionless index of direct hazard; high RHI values indicate high risk of substantial mortality in exposed populations.

ACUTE EXPOSURE MODEL

$$\mathrm{RHI} = \frac{\mathrm{AR} \times \text{Reciprocal of first order rate constants of removal and transfer from water}}{V_{mixing} \times LC_{50}t} \qquad (1)$$

where AR is the recommended application rate for the insecticide in g ha^{-1}; the model accounts for pond surface area and computes loading based on 100% of applied quantities (mg); removal constants ($k_{photolysis} + k_{volatilization} + k_{hydrolysis}$) and transfer rate constants ($k_{suspended\ particulates}$ and $k_{sediments}$) are included (day^{-1}); removal by fish is assumed negligible; V_{mixing} is equivalent to the volume at a specific depth below the surface or the entire volume of a shallow pond and $LC_{50}t$ is the estimated integrated mean lethal exposure in the exposure units (mgL^{-1} days).

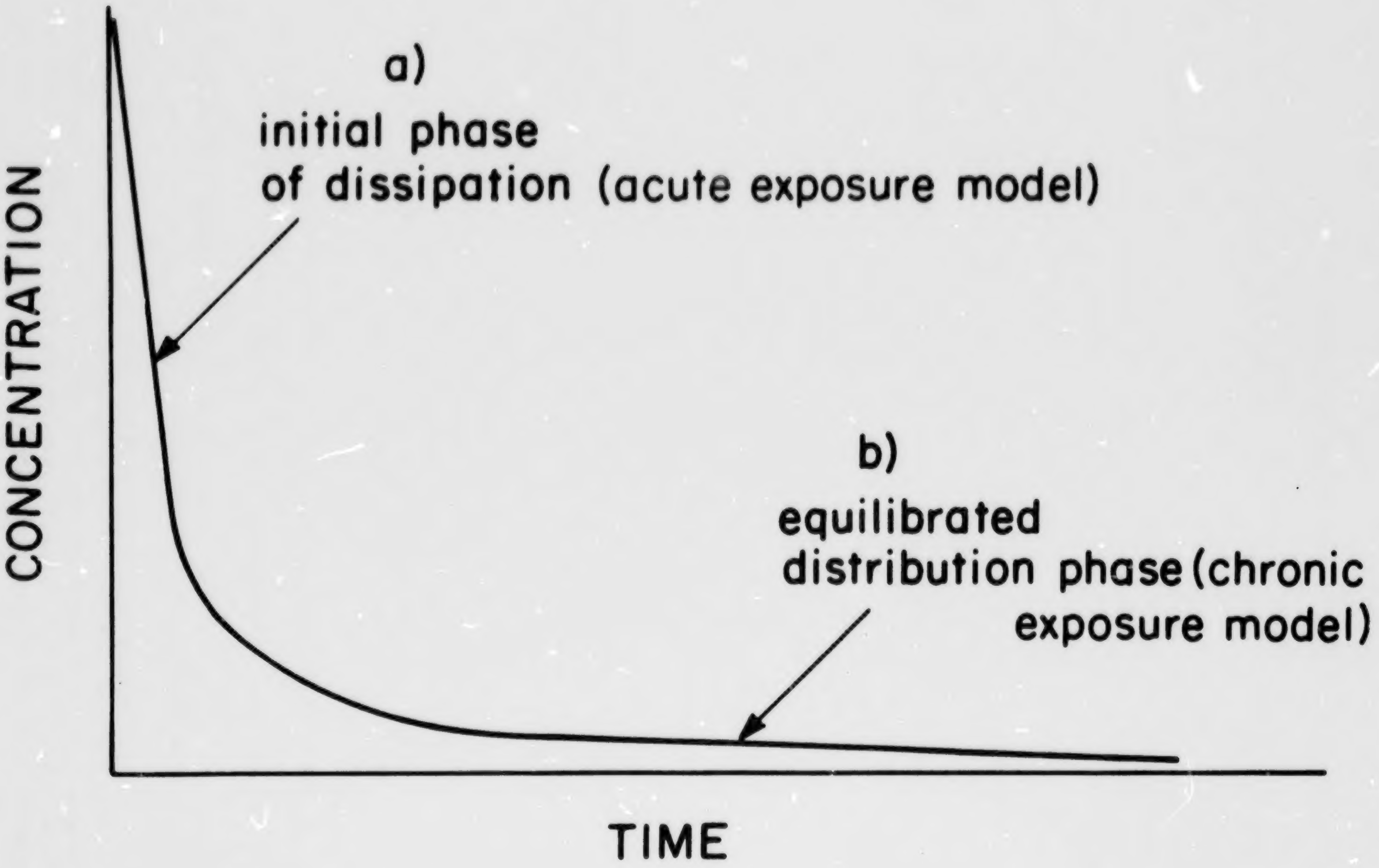
CHRONIC EXPOSURE MODEL

$$\mathrm{RHI} = \frac{F_1 \times \mathrm{AR} \times \mathrm{RC}}{V_{mixing} \times LC_{50}{}^{t}} \qquad (2)$$

where AR, V_{mixing} and $LC_{50}t$ are as defined above; F_1 is the fraction of the insecticide in the water at equilibrium and RC is the residence time of the compound in the aquatic ecosystem.

Theoretical considerations and further development of the hazard models' toxicological and exposure components are presented in detail in the following section to fully acquaint the reader with the procedures and assumptions. Specific application of the general models to hazard assessment in shallow ponds such as prairie potholes is developed.

Figure 6.1. Hypothetical dissipation curve for a single application of a slightly soluble organic insecticide in water of a shallow pond.


a)
initial phase
of dissipation (acute exposure model)
b)
equilibrated
distribution phase (chronic
exposure model)
CONCENTRATION
TIME

6.3.4 Description of the exposure model component

The PERSISTENCE MODEL (NRCC 1981) involves a generalized view of the aquatic environment with four compartments (water, sediment, fish, catchall); it is similar to the FUGACITY MODEL (MacKay & Patterson 1982) and EXAMS (Burns et al. 1982). A schematic of the four compartment model is presented in Figure 6.2. First-order transfer and removal coefficients, and partition coefficients between each compartment and water, are indicated. The catchall compartment is included to consider sorptive components such as algae and suspended solids.

The following constraints and assumptions are used:

1. Only well-characterized chemical and physical processes are included in the analysis of pollutant fate. The intent of the model is to allow the user to determine what is predictable using valid data, and then to predict the rate at which other processes must occur to significantly influence pollutant behaviour patterns. This allows the user to input a minimum of valid data to predict the fate of a pollutant when its retention in the system is maximized or when it has accumulated to a critical extent in a compartment with sensitive species (worst case situations). The model has a bias towards predicting maximum persistence. This is appropriate for pesticide screening and, as additional pieces of valid data become available, refinement in fate predictions are possible.

2. Hydrodynamic properties and settling effects, which cause dilution and redeposition of the compound but do not affect degradation patterns, are not included. This constraint may lead to an overestimate of soluble concentrations but the model is again conservative.

3. The size of the compartments are assumed to remain constant (Table 6.10) and the rate of mixing of the pollutant is rapid relative to the rates of all other processes. This should provide a basis for the determination of relative behaviour patterns. The PERSISTENCE MODEL redefines the various compartments to mimic specific environments.

4. For comparative purposes, all processes are assumed to follow simple kinetics and to be approximated by first-order or pseudo-first-order rate constants.

5. It is assumed that equilibrium will eventually describe the distribution of the pollutant.

The PERSISTENCE MODEL predicts the distribution of a pesticide in a shallow pond and the relative importance of the various removal processes using the input data illustrated in Table 6.11. In many cases, data are not available for all input parameters. A number of data and parameter constraints are discussed in the table. Details of the specific

Figure 6.2. Schematic of the four-compartment model. k_{12}, k_{21}, k_{13}, k_{31}, k_{14}, k_{41} = first-order transfer rate constants. K_{CA}, K_S, K_F = partition coefficients between each compartment and water. k_{01}, k_{02}, k_{03}, k_{04} = first-order removal rate constants (from NRCC 1981).

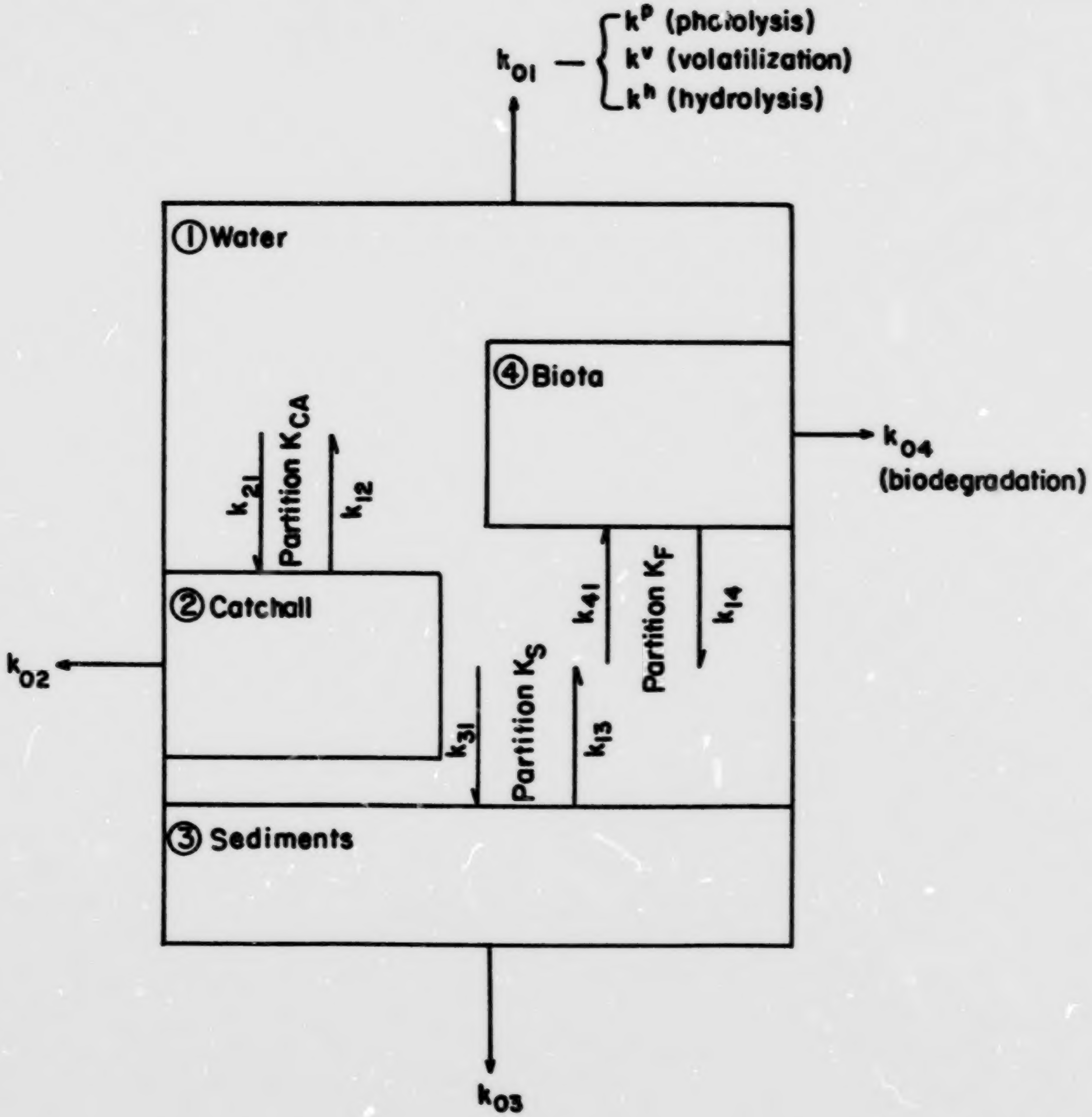
k_{01} — k^p (photolysis)
k^v (volatilization)
k^h (hydrolysis)
① Water
④ Biota
k_{04} (biodegradation)
k_{21}
Partition K_{CA}
k_{12}
② Catchall
k_{02}
k_{41}
Partition K_F
k_{14}
k_{31}
Partition K_S
k_{13}
③ Sediments
k_{03}

Table 6.10. Definition of the standard pond system.

	Pond
Latitude (°N)	45
Temperature (°C)	20
Water	
Surface area (m^2)	4550
Mean depth (m)	1.00
Volume (L)	3.7×10^6
Suspended solids (ppm)	50
Non-pollutant light attenuation coefficient	high[a]
Sediment	
Weight (g)	7.3×10^7
Effective depth for sediment absorption (cm)	1
Organic matter (%)	10
Biota[b]	
Number of fish	185
Average weight of each fish (g)	100

[a] Light absorption for six polluted water bodies (see Zepp & Cline 1977)

[b] Fish are not considered in the analysis of shallow prairie potholes where they are unlikely inhabitants.

Table 6.11. Constraints on input data and effects of input parameters on pollutant dynamics. (from NRCC, 1981)

Parameter		Effect on solution
Solubility (S) or Octanol-water partition coefficient (K_{OW})	The program estimates S from K_{OW}, or K_{OW} from S if no K_{OW} input, at the specified system temperature. Input values must be compatible with system temperature.	$\uparrow K_{OW} \rightarrow \uparrow$ BCF $\uparrow K_S$ usually $\uparrow$ RC
Vapor pressure (vP)	Input value must be compatible with system temperature, i.e. measurement must be made for the same phase in which chemical will exist in the system. A vP for liquid phase may be estimated from the boiling temperature (BP, °Kelvin) where pressure, unless specified, is 760 mm. $\ln \frac{vP_2}{vP_1} = \frac{\varepsilon}{R} (T_2^{-1} - T_1^{-1})$ where R is the gas constant and $\frac{\varepsilon}{BP} = -22$ cal deg^{-1} mol^{-1} If necessary, k^v may be hand calculated and input to the program (Chapter 6).	$\uparrow vP \rightarrow \uparrow k^v$ $\downarrow$ RC
Molecular weight (MW)	It is important to enter correct molecular weight. (Problems will only arise when dealing with mixtures of compounds, e.g. toxaphene.)	$\uparrow MW \rightarrow \downarrow k^v$ $\uparrow$ RC
Quantum yield (Q)	Measurements must be made in an appropriate solvent in the presence of oxygen. Illumination must be at λ >290 nm; Q for λ >290 is constant.	$\uparrow Q \rightarrow \uparrow k^P$ $\downarrow$ RC
Absorbence profile (E_λ)	Extinction coefficients for pollutant must be entered for decadic wavelengths in range 290-400 nm. Values must be determined in an appropriate solvent relevant to water. kP will be underestimated if pollutant absorbs light of λ >400 nm. If Q = 0, E_λ values may be entered as zero.	$\uparrow E_\lambda \rightarrow \uparrow kP$ $\downarrow$ RC
Fraction biodegradable (X)	A value ($0 \leq X \leq 1$) is input to estimate the fraction of pollutant that is biodegradable. This parameter may usually be entered as zero, with little effect on the system.	$\uparrow X \rightarrow \uparrow k_{04}$ $\downarrow$ RC
Hydrolysis rate constant (k^h)	Input value must be a pseudo-first-order, experimentally determined, rate constant for hydrolysis, specific for a given pH and temperature compatible with model system. k^h may include any first-order or pseudo-first-order removal process specific to the water compartment.	$\uparrow k^h \rightarrow \downarrow$ RC
Microbial and other degradation rate constants (k_{02}, k_{03})	Laboratory measurements of microbial degradation are inadequate (see Section 4.2). If reliable values are not available then k_{02} and k_{03} (k_{ox}) can be estimated so as to cause a percent change in RC using CRIT %.	$\uparrow k_{ox} \rightarrow \downarrow$ RC
Critical Percentage (Crit %)	See Table 3-1. This value specifies removal rate constants, specific to sediment and suspended solids, that cause a user-entered percent change in RC.	$\uparrow$(Crit %)$\rightarrow \downarrow$ RC
Melting point (MP)	Value may be entered as zero if compound is a liquid at system temperature. If compound is a solid, MP is used to compute the liquid solubility. In turn, vP may be calculated by hand and the program rerun.	---

Table 6.12. Data from the literature describing the physico-chemical properties and degradation contents of these insecticide compounds to be analysed in the aquatic persistence model.

Insecticide	Melting Point^a C°	Molecular Weight	Vapour Pressure mm Hg	Reference	Octanol-Water Partition Coef.	Reference	Solubility mg L^{-1}	Reference	Quantum Yield	Reference	Hydrolysis Rate Constant day^{-1}	Reference
Synthetic Pyrethroids												
cypermethrin	0	416	$1.2x10^{-7}$ at 80°C	Breese & Highwood 1977	29,500	Coats & O'Donnell-Jeffery 1979	0.009	FAO/WHO 1980	0.12	Miyamoto & Mikami 1983	$1.1x10^{-1}$ at pH 6 at 75°C	Greyson 1975 in NRCC 1986
			$1.2x10^{-8}$ at 30°C	(estimated assuming the heat of vaporisation is similar to that for permethrin)	1,585,000	Elliott & Janes 1978	0.041	Coats & O'Donnell-Jeffery 1979			$3.16x10^{-1}$ at pH 8	
					2,100,000	Crossland 1982	0.1	Elliott & Janes 1978				
			$3.0x10^{-8}$	Elliott & Janes 1978							$3.4x10^{-3}$ at pH 6^b $9.9x10^{-3}$ at pH 8^b	
deltamethrin			$1.5x10^{-8}$ at 25°C	Roussel-Uclaf (unpubl.)				Worthing & Walker 1983				(assumed to be equivalent to rate constants estimated for cypermethrin)
	$1 x 10^{-9}$			Elliott & Janes 1978	2,700,000	Roussel-Uclaf (unpubl.)	0.002	Elliott & Janes 1978	0.28	Miyamoto & Mikami 1983	$3.4x10^{-3}$ at pH 6 $9.9x10^{-3}$ at pH 8	
	101	505.2	$3 x 10^{-10}$ at 25°C	Roussel-Uclaf 1982	2,512,000	Elliott & Janes 1978	0.1					
fenvalerate	0	420	$2.8x10^{-7}$ at 25°C	Yoshioka 1978	26,500	Coats & O'Donnell-Jeffery 1979	0.002	FAO/WHO 1980	0.08	Miyamoto & Mikami 1983	$2.33x10^{-2}$ at pH 6 at 75°C $2.36x10^{-1}$ at pH 8	Greyson 1975 in NRCC 1986
			$1.44-2.38x10^{-7}$ at 20°C $1x10^{-9}$	Loubaresse et al. 1977 Ohno et al. 1976	103,000 1,585,000	Yoshioka 1978 Schimmel et al. 1983	0.01 0.085	Ohno et al. 1976 Coats & O'Donnell-Jeffery 1979				
					6,310,000	Ohno et al. 1976	0.02-0.32 0.31 at 23°C	Ohkawa et al. 1978 Loubaresse et al. 1977			$7.28x10^{-4}$ at pH 6^b $7.38x10^{-3}$ at pH 8^b	

Table 6.12 (cont'd)

Insecticide	Melting Point[a] °C	Molecular Weight	Vapour Pressure mm Hg	Reference	Octanol-Water Partition Coef.	Reference	Solubility mg L^{-1}	Reference	Quantum Yield	Reference	Hydrolysis Rate Constant day^{-1}	Reference
permethrin	0	391	1.0x10^{-7}	Elliott & James 1978	3,060	Coats & O'Donnell-Jeffery 1979	0.02	Martin & Worthing 1977	0.51	Miyamoto & Mikami 1983	2.0-5.5x10^{-3} at pH 9 at 25°C 0 at pH 6	Allsup 1976 *in* NRCC 1986
							0.03	Elliott & James 1978				
					170,000	Lockhart *et al.* 1984	0.04	Coats & O'Donnell-Jeffery 1979			1.26x10^{-2} at pH 4 at 22-27°C 1.08x10^{-2} at pH 7 (cis isomer) 7x10^{-3} at pH 9	
					3,162,000	Schimmel *et al.* 1983	0.05	Schimmel *et al.* 1983				
							0.07	Fujie 1975 *in* NRCC 1986				
											1.14x10^{-2} at pH 4 at 22-27°C 7.14x10^{-3} at pH 7 (trans isomer) 5.33x10^{-3} at pH 9	Fujie 1975 *in* NRCC 1986
											2.0x10^{-2} at pH 6 at 75°C 4.0x10^{-2} at pH 8	Fujie 1975 *in* NRCC 1986
											3x10^{-7} at pH 5^{b} 3x10^{-3} at pH 9^{b}	Grayson 1975 *in* NRCC 1986

Table 6.12 (cont'd)

Insecticide	Melting Point^a C°	Molecular Weight	Vapour Pressure mm Hg	Reference	Octanol-Water Partition Coef.	Reference	Solubility mg L^{-1}	Reference	Quantum Yield	Reference	Hydrolysis Rate Constant day^{-1}	Reference
Other Insecticides												
azinphos methyl	73	317.3	3.8×10^{-4} at 20°C	Martin & Worthing 1977	NA^c		29 at 25°C 33 at 25°C 22 at 20°C^e 21	Chemagro Research Division 1976 Spencer 1982 Bowman & Sans 1983	NA		1.26×10^{-2} at pH 7.4 at 20°C 5.41×10^{-2} at pH 8.8 0 at pH 5^b 6.1×10^{-2} at pH 9^b 8×10^{-3} at pH 7.8 at 25°C 2.5×10^{-2} at pH 8.6 0 at pH 5^b 3.3×10^{-2} at pH 9 4×10^{-2} at pH 5 at 20°C 1.5×10^{-1} at pH 9	de Boer 1979 de Boer 1979 Bayer A.G. 1978 in de Boer 1979
carbaryl	142	201	5×10^{-3} at 26°C	Verschueren 1983	81 197 225	O'Brien & Dannelly 1965 Kanazawa 1981 cf. Lyman et al. 1982	34 at 20-25°C 40 at 30°C 82.6 column method 190 RPHPLC 104	Kanazawa 1981 Verschueren 1983 Swann et al. 1983 Bowman & Sans 1983	.006	Wolfe et al. 1978	6.6×10^{-6} at pH 5 at 27°C 4.6×10^{-2} at pH 7 4.6 at pH 9 1.33×10^{-4} at pH 5 at 20°C^b 1.33 at pH 9^b 6.6×10^{-2} at pH 7 3.8×10^{-1} at pH 8 6.9 at pH 9^b	Wolfe et al. 1978 Aly & El Dib 1971

Table 6.12 (cont'd)

Insecticide	Melting Point[a] C°	Molecular Weight	Vapour Pressure mm Hg	Reference	Octanol-Water Partition Coef.	Reference	Solubility mg L^{-1}	Reference	Quantum Yield	Reference	Hydrolysis Rate Constant day^{-1}	Reference
carbofuran	150-152	221.3	2x10^{-5} at 33°C 3.4x10^{-6} at 25°C 6.84x10^{-7} at 20°C^{b}	Verschueren 1983 NRCC 1979	17 60 200 759	NRCC 1979 cf. Lyman et al. 1982 Verschueren 1983 Bowman & Sans 1979	700 at 25°C 320 415 400 column method 670 RPHPLC	Verschueren 1983 Bowman & Sans 1979 cf. Lyman et al. 1982 Swann et al. 1983	NA		3.4x10^{-6} at pH 5 at 25°C 1.7x10^{-3} at pH 6 1.73x10^{-1} at pH 8 1.19 at pH 9	NRCC 1979
chlorpyrifos	41.5-43.5	350.5	1.87x10^{-5} at 20°C 1.87x10^{-5} at 25°C 8.7x10^{-5} at 35°C 4.62x10^{-6} at 20°C^{b}	Verschueren 1983 NRCC 1978a	66,000 129,000	NRCC 1978a Freed et al. 1979	0.4 at 23°C 2.0 at 25°C 1.07 column method 0.42 RPHPLC 0.73	NRCC 1978a Gray 1965 Swann et al. 1983 Bowman & Sans 1983	NA		1.1x10^{-2} at pH 4.7 at 25°C 1.19x10^{-2} at pH 6.9 3.0x10^{-2} at pH 8.1 2.1x10^{-2} at pH 5^{b} 1.3x10^{-2} at pH 6 3.85x10^{-2} at pH 8 4.33x10^{-2} at pH 9 6x10^{-3} at pH 6.1 at 20°C 1.3x10^{-2} at pH 7.4 8.9x10^{-3} at pH 7 at 25°C 1.46x10^{-2} at pH 9	Meikle 1972 Brown 1975 in NRCC 1986 Freed et al. 1979 Macalady & Wolfe 1983
diazinon	83-84	304.36	1.4x10^{-3} at 20°C	Martin & Worthing 1977	1,386	Kanazawa 1981	40 40.5 at 20-25°C 68.8	Martin & Worthing 1977 Kanazawa 1981 Bowman & Sans 1983	NA		2.2x10^{-2} at pH 3.1 at 20°C 3.7x10^{-3} at pH 7.4 5.0x10^{-3} at pH 9	Gomaa et al. 1969

Table 6.12 (cont'd)

Insecticide	Melting Point[a] C°	Molecular Weight	Vapour Pressure mm Hg	Reference	Octanol-Water Partition Coef.	Reference	Solubility mg L^{-1}	Reference	Quantum Yield	Reference	Hydrolysis Rate Constant day^{-1}	Reference
dimethoate	48-52	229.3	$2.5x10^{-5}$ at 25°C $4.3x10^{-5}$ at 30°C $8.64x10^{-5}$ at 35°C $7.95x10^{-6}$ at 20°C[b] $8.2x10^{-6}$ at 25°C $7.3x10^{-6}$ at 20°C $2.5x10^{-2}$ at 25°C	USDA 1979 Wagner & Prehoe 1976 Martin & Worthing 1977 Verschueren 1983	513 508	Verschueren 1983 USDA 1979	25,000	Verschueren 1983 Bowman & Sans 1979 cf. Lyman et al. 1982 Swann et al. 1983	NA		$3.4x10^{-4}$ at pH 3 at 25°C $1.7x10^{-3}$ at pH 6 $1.73x10^{-1}$ at pH 8 1.19 at pH 9	NRCC 1979
malathion	2.9-6.1	330.4	$4x10^{-5}$ at 20°C $4x10^{-5}$ at 30°C $1.25x10^{-4}$ at 20-25°C	Verschueren 1983 Martin & Worthing 1977 Melnikov 1971	781	Freed et al. 1979	145 at 20°C 143	NRC 1978/Burst 1964 Gray 1983 Swann et al. 1983 Bowman & Sans 1983	NA		$1.1x10^{-2}$ at pH 6.7 at 25°C $1.19x10^{-2}$ at pH 6.9 $5.0x10^{-2}$ at pH 8.1 $2.1x10^{-2}$ at pH 3^{b} $1.3x10^{-2}$ at pH 4 $3.85x10^{-2}$ at pH 6 $4.33x10^{-2}$ at pH 9 $6x10^{-3}$ at pH 6.1 at 20°C $1.3x10^{-2}$ at pH 7.4 $8.9x10^{-3}$ at pH 7 at 25°C $1.44x10^{-2}$ at pH 9	Meikle 1972 Brown 1975 in NRCC 1986 Freed et al. 1979 Macalady & Wolfe 1983
methoxychlor	89	345.5	$3.5x10^{-5}$ at 31°C $1.24x10^{-6}$ at 25°C $6.4x10^{-7}$ at 25°C	Cf. Biticy et al. 1981 J.R. Roberts NRCC (pers. comm.) T. Bidleman, University of South Carolina (pers. comm.)	1,122 2,061 4,786 2,500 19,950 47,500 120,000	Coats et al. 1979 Kapoor et al. 1973 Kapoor et al. 1970 Wolfe et al. 1977b Veith et al. 1979 Kenaga 1980 Karickhoff et al. 1979	0.003 0.04 at 24°C 0.1 at 25°C 0.12 at 25°C 0.62	Martin & Worthing 1977 Kenaseuo 1981 Bowman & Sans 1983	NA		$2.2x10^{-2}$ at pH 5.0 at 20°C $3.7x10^{-3}$ at pH 7.4 $5.0x10^{-3}$ at pH 9	Gomaa et al. 1969

Table 6.12 (cont'd)

Insecticide	Melting Point[a] C°	Molecular Weight gram-molecular weight	Vapour Pressure mm Hg	Reference	Octanol-Water Partition Coef.	Reference	Solubility mg L^{-1}	Reference	Quantum Yield	Reference	Hydrolysis Rate Constant day^{-1}	Reference
phosmet	72-72.7	317.3	1x10^{-3} at 50°C	Freed *et al.* 1979 Chiou *et al.* 1977	677	Freed *et al.* 1979	25 at 25°C	Melnikov 1971	NA		9.9x10^{-2} at pH 6.5 at 20°C 2.34 at pH 7.4 9x10^{-2} at pH 5[b] 6.3 at pH 9[b]	Freed *et al.* 1979
							24.4	Bowman & Sans 1983				

[a] Melting point entered as 0 if the compound is a liquid at system temperature, 20°C
[b] Values extrapolated to meet the pH and temperature conditions of the model
[c] Data not available

calculations used in the model predictions are discussed in Part I of NRCC (1981). Outputs of the algebraic solution include 1) partition coefficients between each compartment and water, 2) first-order transfer and removal rate constants for each compartment, 3) retentive capacity of the system, 4) half-life of the pollutant in the system, 5) fractional retention in each compartment and 6) fractional removal by each process. The outputs of the graphic or tabular solution describe the dynamics of distribution and decay of a single pulse exposure. The one output requiring further definition is retentive capacity (RC). In mathematical terms, retentive capacity is

$$RC = \frac{dQ_t^{\;-1}}{Q_{t}dt} \tag{3}$$

or simply the inverse of the specific removal rate of the pollutant from the system at any time, t, where Q_t is the total amount of the pollutant in the system at any time. Retentive capacity therefore describes the residence time of a chemical in an aquatic ecosystem as a function of characteristics of the system and characteristics of the chemical (see Roberts & Marshall 1980).

A typical dissipation curve for a single input of an organic insecticide to a shallow pond was illustrated in Figure 6.1. It is clear from the shape of the persistence curve for the compound in water that there are at least two distinct phases of dissipation. An initial rapid loss (section a) controlled primarily by transfer processes; the distribution of the chemical within the pond ecosystem has not yet equilibrated. Prior to insecticide equilibration among compartments, the reciprocal of the summation of first order transfer and removal rate constants for the water compartment provide a measure of retention. Of particular importance are transfers to suspended particulates and benthic sediments. These are expected to be diffusion controlled processes dependent on the size and composition of the particulate compartments but independent of the physicochemical properties of organic pollutants. For the pond described in Table 6.10, the transfer constant to suspended particulates (k_{21}) is assigned a value of 0.005 days^{-1} and the transfer constant for the benthic sediment is (k_{31}) equal to 0.2 days^{-1}. Transfers to macrophyte vegetation are not defined in the PERSISTENCE MODEL. The importance of plant removal of organic contaminant from aquatic systems is recognized but, at present, little quantitative information is available upon which to base the calculation of transfer constants for macrophyte uptake and retention. In initial estimates of persistence, transfer to macrophytes was assumed to be represented within the general catchall compartment described earlier. This is a conservative assumption in that predictions of persistence in the water column are biased toward a maximum but could be somewhat lowered with model refinements incorporating more accurate calculations of transfers to higher aquatic plants. Removal processes which influence persistence include hydrolysis (K_h), photolysis (K_p) and volatilization (K_v). Biodegradation, which certainly influences the persistence of organic pollutants in the longer term, is considered negligible during the period of initial dissipation.

Under conditions of equilibration in the distribution of an organic pollutant among environmental compartments (Figure 6.1, section b), the persistence of the compound in the water is predicted by its fractional retention in the water column (F_1) multiplied by the calculated retentive capacity (RC) (days). The rate of dissipation during this phase is slow and the estimated environmental concentrations are generally low. Losses are largely controlled by chemical and biological degradation processes. Although the NRCC PERSISTENCE MODEL is designed to predict aquatic environmental concentrations of pollutants under both pre- and post-equilibration conditions, the lack of reliable data to input in the calculation of biodegradation losses leads to the assumption of their negligible contribution to the dissipation of pollutants. This can substantially reduce the effectiveness of the post-equilibration simulations in predicting aquatic persistence. If the importance of microbial transformations is great relative to other removal processes, model predictions of dissolved concentrations will be high.

Because of the differences in the relative contribution of processes controlling persistence and the vast differences in the rate of compound loss within the two dissipation phases, no single exposure model is, in itself, adequate to predict expected environmental concentrations throughout time. Exposure models specific to the initial phase

$\left| \dfrac{AR}{(k_{21} + k_{31} + k_h + k_p + k_v)} \right|$ and to the equilibrated distribution phase

$[F_1RC]$ can provide a reasonable base for the estimate of expected concentration in risk analysis calculations during two quite different but critical periods of exposure. An estimated high level of an insecticide in water initially following application is an indication of short-term acute exposure conditions. An extended period of low-level residues of the insecticide in water is representative of chronic exposure conditions.

At present, no attempt has been made to quantify the uncertainty in estimates of persistence for hazard calculations, although it is possible to estimate magnitudes of errors associated with uncertain estimates of physico-chemical properties of contaminants (e.g. Bartell et al. 1983).

6.3.5 Description of the toxicity model component

The definition of a toxicity model requires the selection of an appropriate comparable toxicological benchmark and the analysis of available test data to determine whether the effect appears to be a function of estimated integrated exposure or estimated maximum exposure.

Several possible toxicological benchmarks which could be incorporated into relative hazard calculations are available from laboratory toxicity tests. Among available water quality criteria are the EC_{50} (concentration causing a designated effect on 50% of the test organisms within a specific exposure period), the LC_{50} (concentration killing 50% of the test organisms within a specific exposure period), the TLm (median tolerance level or concentration at which 50% of the test organism survive within a specific exposure period), the NOEL (no observed effect level defined as the highest test concentration at which no chronic effects are observed) and the MATC (maximum allowable toxic concentration range defined by the highest value showing no detectable harmful effect and the lowest value displaying some observable effect). EC_{50} and LC_{50} define the acute toxic response and MATC and NOEL define the boundary for chronic toxicity. The most widely reported toxicological benchmark is the LC_{50}, and, as LC_{50} data sets collected under standardized conditions are generally available for various species and pesticides, this benchmark offers the best opportunity for broadscale data comparisons. Additionally, Tucker and Lietzke (1979) argued that, because indices of sublethal effect tend to occur at levels rarely much less than a quarter to a sixth of the level producing a comparable percent mortality in equivalent tests, LC_{50} is a clear, consistent indicator of potential impact (even for chronic exposure). There are few reported MATC or NOEL values for aquatic species exposed to pesticides which reduces the utility of these benchmarks in hazard models at the present time.

The theory of the concentration-response relationship producing LC_{50} values assumes 1) death is a positive function of exposure, 2) response magnitudes are proportional to the logarithm of concentration and 3) randomly selected test animals are normally distributed in their sensitivity to a toxicant. These assumptions are largely met in most toxicity tests if close attention is paid to controlling variables which influence contact and absorption of the toxic compound. For example, carriers, surfactants and formulation can greatly alter the adsorption and apparent toxicity of pesticides (e.g. Stratton et al. 1980). Alteration of the temperature or pH of the test water can also alter adsorption and apparent toxicity (e.g. Eisler 1970). In addition, the loss of pesticide from the test vessel through volatilization, sorption or bioaccumulation by over-dense test populations, or the rapid transformation of the compound through biological or chemical processes, can result in a reduction in real toxicant concentrations below nominal dosage levels either invalidating the test or grossly underestimating the toxicity. The numerous examples of experimental and environmental variables influencing toxicity test results (see Tucker & Lietzke 1979 for discussion) suggest that the comparative toxicology of pesticides to aquatic invertebrates must be obtained from experimental data sets where methods are well defined and reasonably comparable.

6.3.5.1 Integrated exposure model

Theoretically, for organisms exposed at a single concentration of a toxicant, lethal toxicity is usually viewed as a function of percent mortality and survival time. At each higher concentration the organisms will die more rapidly. Results can be plotted on a three-dimensional graph of concentration, percent mortality and survival time (see Skidmore 1974 for discussion). Discounting any lag period or delayed effects, which would occur if adsorption and transport to the site of action were other than first-order processes, the relationship between median lethal concentration of the test compound and duration of exposure can be expressed as $LC_{50} \times t = k$ where t is the duration of exposure and k is a toxicity constant which approximates the integrated median lethal exposure. This expression implies that a low concentration rate over a long treatment period can produce the same effect as a high concentration over a short period. Although the fixed relationship between lethality and time does not hold for all pesticides and species, it can serve as an appropriate model for compound comparisons. Correlation coefficients for the regression of LC_{50} values against test duration for toxicity tests conducted on the amphipod, Gammarus lacustris, and the stonefly, Pteronarcys californica, for seven insecticides were greater than $r = -.87$ (see data from Sanders and Cope 1968 and Sanders 1969 in Tables 6.3 and 6.6, respectively). If doubling the exposure time cuts the median lethal concentration in half, the model is one of perfect chronicity. Although this model may not accurately fit all toxicity test values, it is not unusual to see a 24 h LC_{50} which is twice the value of the 48 h LC_{50} which is, in turn, twice the toxicity reported for 96 h exposure if there is no substantial loss or transformation of the toxicants throughout exposure. Table 6.13 illustrates the narrow range of chronicity ratios for Gammarus lacustris populations exposed under similar test conditions to a number of insecticides. The chronicity ratios for pyrethroid compounds cannot be calculated from available data.

A number of studies have found correlations between reported LC_{50}'s and the estimated integrated exposure. Allison (1977) suggested that an exposure-unit (mgL^{-1} day) may have value in estimating the environmental impact of pesticides on fish over a range of exposure conditions. LC_{50} data for fish exposed to chlorpyrifos also suggest the applicability of the median lethal exposure model to explain toxicity: reported LC_{50} values were correlated with the net accumulation of chlorpyrifos in fish, but LC_{50} values and maximum concentrations in fish did not demonstrate any apparent trend (NRCC 1981). The integrated model is particularly appealing because the value of $LC_{50}t$ for short exposure periods approaches that of the alternative toxicity model based on maximum concentration.

6.3.5.2 Maximum concentration model

The maximum concentration model is based on an organism response which occurs only if the threshold concentration is exceeded by a massive dose which overwhelms its normal steady state kinetics for the compound. This is an acute model of an all or none response. When the threshold concentration is exceeded, the toxic mechanism is initiated and the response process becomes irreversible. Below this critical dose, mechanistic response is reversible. If threshold concentrations are not bioaccumulated, no period of extended exposure below the threshold will ever produce a response. Accumulation of the toxic compound at the site of action is therefore generally unrelated to duration of exposure (beyond the lag period necessary to reach steady state) but is correlated with the frequency with which the critical concentration in the environment is exceeded. Accumulated storage of organochlorine compounds can act as an alternative pathway to exceed the estimated maximum concentration. There are few clear examples of the applicability of this model to explain the toxic effects of pesticides on aquatic organisms. No examples were found for the aquatic invertebrate species examined in Section 6.1 and there were few correlations of LC_{50} values with estimated maximum exposure concentrations for fish species exposed to the same insecticides. Eisler (1970) presented data which suggest that four estuarine fish respond to a maximum concentration when exposed to methoxychlor. It should be noted, however, that the toxicity test data, as often reported, provides

Table 6.13 Toxicity test chronicity for *Gammarus lacustris* exposed to insecticides[a]

Insecticide	24 h LC_{50}: 48 h LC_{50}	48 h LC_{50}: 96 h LC_{50}
azinphos methyl	2.24	1.67
carbaryl	1.82	1.38
chlorpyrifos	1.88	3.63
diazinon	1.60	2.50
dimethoate	2.25	2.00
malathion	2.11	1.80
methoxychlor	2.23	1.63

[a] Data from Sanders 1969

insufficient information on which to base conclusions on the suitability of this toxicity model.

6.3.5.3. Choice of test species

Because of the strong correlation of the gammarid and stonefly toxicity data with the estimated median lethal exposure model, and the convergence of it with the estimated maximum concentration model during the period immediately following contamination when concentrations are high, the integrated exposure model will be used in relative hazard evaluations.

The availability of appropriate data sets for application of the integrated median lethal exposure model is limited, particularly for invertebrate species important to the feeding of ducklings and laying hens. There are comparative LC_{50} data for the amphipod genus, Gammarus (e.g. Sanders 1969, 1972), which is a choice prey in the early diet of diving duck species (see Section 2) and is highly susceptible to chemical pollutants. Sloof (1983) reported that Gammarus pulex was the most sensitive of eleven macroinvertebrate species exposed to 15 chemicals in short-term laboratory tests. Toxicity test protocols for gammarids are well described (reviewed by Arthur 1980) and the reporting of LC_{50} data for amphipods in pesticide registration submissions is common but not universal.

A broad base of toxicity data is available for the midge family, Chironomidae, a second group important as prey for both diving and dabbling ducks. Short-term LC_{50} values for chironomids come largely from efficacy studies with potential pest species (e.g. Ali & Mulla 1978a, 1980) and chironomids appear to be relatively less sensitive to toxicants than amphipod species (see Tables 6.1, 6.4, 6.6 and Sloof 1983). Standard toxicity test procedures for midge species have been described (Anderson 1980).

Efficacy studies also provide a broad comparative data base for mosquito larvae. As noted earlier (Section 2.5), mosquitoes make up a small proportion of duck diets, although they can be important prey in isolated cases when larval density is high.

The only internationally recognized invertebrate taxa for toxicity testing is Daphnia (OECD 1981). Cladocerans appear to be largely incidental selections as prey by ducks, with the exception of the Northern Shoveler. However, Daphnia is perhaps the most widely chosen aquatic invertebrate for toxicity testing, test protocol are highly developed (Buikema et al. 1980, OECD 1981) and Daphnia is often the only zooplankton genera for which comparative pesticide test data are available (e.g. Sanders & Cope 1966).

Comparative LC_{50} data for insecticides gathered under the same test conditions are also available for the stonefly Pteronarcys californica (Sanders & Cope 1968). This aquatic insect is relatively tolerant of insecticides and does not contribute to the diet of ducks, but it may be representative of other large-sized insects (mayflies and caddisflies) which are important prey.

It is important to note that no one taxa met all the criteria of an ideal invertebrate species for our relative hazard model, including availability of comparative data, duration-effect correspondence, dietary importance and sensitivity to insecticides. Bothersome problems also include the inconsistency in the selection of a standard test organism for baseline studies. The work of Sloof (1983), however, indicates that the maximum difference in relative tolerence among species is likely to be relatively low: less than a factor of five for 15 toxicants tested on 11 invertebrates. Hazard analysis, can, therefore, be based on toxicity data from the most sensitive species, the most ecologically relevant species, or on the range of values across species for which data is available.

The data summarized in Table 6.14 show the range of integrated median lethal exposure values for Daphnia magna, Gammarus lacustris, Pteronarcys californica, Chironomus spp. and Culex spp. calculated for one to four day exposure periods. For Gammarus and Pteronarcys the $LC_{50}t$ range is quite narrow. Insufficient data are available to determine the goodness of fit of data summaries for other invertebrate taxa in the integrated exposure model. Within the data sets for individual species, the range of integrated median lethal exposure values for specific insecticides does not exceed a factor of ten. The range among species rarely exceeds 100. The ranges are relatively narrow considering that differences in both exposure period and laboratory procedure are included in the data. Kenaga (1978) stated that the limits of reliability of acute toxicity values are ±1 order of magnitude.

Although the integrated median lethal exposure model appears to adequately explain toxicity data (in those cases where evaluation can be made) for short-term exposure conditions as would occur immediately after pesticide contamination of a pond, the utility of this toxicity model in defining response under conditions of equilibrated distribution of the pesticide is unclear. Kenaga (1982a, b) concluded that there appeared to be no consistent pattern of ratios between acute and chronic toxicity and this was particularly evident for pesticides tested on aquatic organisms.

When chronic exposure data are used in the calculation of $LC_{50}t$, values of integrated median lethal exposure are greater for the longer exposure periods. As there are not sufficient chronic toxicity data from partial or whole life-cycle tests for invertebrate species of interest, no firm conclusions can be drawn on the adequacy of either this model or other possible toxicological models based on chronic toxicity benchmarks (e.g. MATC). As a first approximation, the $LC_{50}t$ model, will be applied to both initial phase and equilibrated phase of relative hazard calculations.

6.3.6 Benchmarks and an index of secondary impacts

The problem of relating the value of the relative hazard index to abundance changes and lags in recovery of exposed communities in natural sloughs can be most readily solved with the field benchmark concept. The relative hazard model is a screening tool and as such does not attempt to generate either an absolute measure of hazard or the probability of hazard occurrence. As an empirical model, it requires the use of indicators, such as well-characterized benchmark chemicals, to link hazard predictions

Table 6.14. Summary of toxicity values for integrated median lethal exposure model reported as range from available data sets.

	$LC_{50}t$ mgL^{-1} day (number of data sets)				
Insecticide	*Daphnia magna*	*Gammarus lacustris*[b]	Aquatic Insects[b]	*Chironomus* spp.	*Culex* spp.
Synthetic Pyrethroids					
cypermethrin	0.002 (1)	0.00006-0.00013 (2)	0.0006Cd (1)	0.0002 (1)	0.00007 (1)
deltamethrin	0.008-0.01 (2)	NA[a]	0.000028Bp (1)	0.00023-0.00029 (2)	0.00002-0.00019 (2)
fenvalerate	0.0066 (1)	0.00012-0.00014ps (3)	0.00039-0.0037E, Pd (2)	0.0042-0.015 (2)	0.0040-0.0047 (2)
permethrin	0.0004 (1)	NA	0.00011Br (1)	NA	0.0014-0.0030 (2)
Other Insecticides					
azinphos methyl	0.0023-0.0040 (2)	0.00056-0.0006 (3)	0.006-0.016, Pc (3)	NA	NA
carbaryl	0.01-0.014 (2)	0.040-0.044 (3)	0.019-0.030, Pc (3)	0.020 (1)	0.480 (1)
carbofuran	0.020 (1)	NA	NA	NA	NA
chlorpyrifos	0.016 (1)	0.00044-0.0008 (3)	0.036-0.050, Pc (3)	0.0005-0.0012 (2)	0.0012-0.0016 (3)
diazinon	0.0025-0.003 (2)	0.800-1.0	0.100-0.155, Pc (3)	NA	0.024-0.031 (2)
dimethoate	5.000-12.800 (3)	0.800-0.900 (3)	0.172-0.510, Pc (3)	NA	NA
malathion	0.0009-0.002 (2)	0.0036-0.004 (3)	0.035-0.040, Pc (3)	0.0030-0.0074 (2)	0.032-0.034 (5)
methoxychlor	0.0037 (1)	0.0026-0.0047 (3)	0.0056-0.030, Pc (3)	0.0065 (1)	0.0089-0.0189 (2)
phosmet	0.0112 (1)	0.003-0.008f (2)	NA	NA	NA

[a] NA - no appropriate data set available

[b] Index calculated from toxicity data for *Gammarus fasciatus* (f), *Gammarus pulex* (p), *Gammarus pseudolimnaeus* (ps), *Cloeon dipterum* (Cd), *Baetis parvus* (Bp), *Baetis rhodani* (Br), *Ephemerella* sp. (E), *Pteronarcys dorsata* (Pd), *Pteronarcys californica* (Pc).

to real-life situations. Extrapolation can be made from the field behaviour and effects of the benchmark to the expected behaviour of the pesticide in similar situations. Thus, with respect to the proper benchmarks, the relative hazard index value of pesticides can be linked to a probable percent depression in population or community biomass and to an expected lag time to achieve some acceptable level of recovery. Of necessity, benchmark compounds must be well characterized with respect to relevant fate processes and adverse effects operative in various field situations. These criteria are not readily met due to the lack of comprehensive field studies on specific insecticides (see Section 6.2). However, for a small number of compounds included in the hazard assessment, there are multiple studies in natural and artificial lentic ecosystems. These compounds are chlorpyrifos, permethrin and, to a lesser extent, cypermethrin, malathion, methoxychlor and carbaryl. The results of the field studies can be used to roughly calibrate relative risk values in terms of both losses in organism abundance and community resilience.

6.3.7 Input data for risk calculations

Specific data input for physico-chemical parameters is given in Appendix 6.1. When a choice of data was available, the selection of an appropriate value was based on the internal agreement among values (e.g. elimination of obvious outliers), the environmental relevance of the data (primarily pH and temperature conditions) and a judgement on the reliability of the experimental methods used to obtain specific values. Hydrolysis rate constants were chosen to represent conditions in an acidic (pH 5) and basic (pH 8-9) pond. The most difficult selections centered on the appropriate octanol-water partition coefficients where there was often a three-fold range of values. For the synthetic pyrethroids, moderate K_{OW} values were selected to agree with National Research Council Canada modelling studies (NRCC 1986). The effects of entering very high or low parameter values will be discussed in Section 6.3.9.

Toxicological data for Daphnia magna, Gammarus lacustris, Pteronarcys californica, Chironomus, Culex or equivalents were input as indicated in Table 6.14. The high and low values of integrated median lethal exposure were used to calculate boundaries on the range of relative risk values to individual taxa.

The recommended application rates for insecticides and the maximum expected initial concentration, based on a worst case of 100% of applied material mixed into pond water, are shown in Table 6.15. Where possible, rates were selected to reflect the control of grasshoppers on cereal crops. If a product was not recommended for the control of grasshoppers, rates for other pests on grain or oil crops were selected. By using the application rates for these insecticides, comparable treatment levels under field conditions were approximated and the hazard ranking generated reflected expected use patterns. However, no attempt was made to simulate actual pesticide loading. For the standard pond model, expected initial insecticide concentrations were calculated under the assumption that compounds reaching the surface are quickly mixed into the entire water column.

Table 6.15. Application rates as input into the relative hazard model normalized when possible to control of grasshoppers on cereal crops.

Insecticide	Recommended Application Rate[a] g AI ha^{-1}	Expected Initial Concentration in Pond Water[b] $\mu g\ L^{-1}$
Synthetic Pyrethroids		
cypermethrin	28	3
deltamethrin	7.5	1
fenvalerate	97.5	12
permethrin	140	17
Other Insecticides		
azinphos methyl	420	52
carbaryl	1100	135
carbofuran	140	17
chlorpyrifos	560	69
diazinon	550	68
dimethoate	490	60
malathion	840	103
methoxychlor	1500	184
phosmet	1125	138

[a] Recommended application rates based on those provide in insect control pamphlets for Alberta, Manitoba and Saskatchewan

[b] Concentration calculated by multiplying recommended application rate times surface area/volume of the model pond.

6.3.8 Results from relative hazard analysis

6.3.8.1 Persistence

The computer-predicted distribution and degradation patterns of selected synthetic pyrethroid insecticides are summarized in Table 6.16. These insecticides are predominantly sequestered in benthic sediment (<92%) with less than 5% of the total load in the water under conditions of equilibrated distribution. Fenvalerate and permethrin are primarily removed via volatilization, however, losses are expected to be slow because most of the insecticide will be in the sediments. Therefore, retentive capacities (RC) of approximately 850-17,000 days are predicted for these compounds. In contrast, hydrolysis is the main route of removal of cypermethrin from a shallow pond ecosystem. Because hydrolysis rates are slow, the retentive capacity of this compound is predicted to be greater than 1,600 days. Assuming that the hydrolysis rates of deltamethrin are similar to those of permethrin, deltamethrin is primarily hydrolyzed and shows the longest RC of the four synthetic pyrethroids (>10,000 days).

The persistence of the pyrethroids is affected by pH, with degradation more rapid in alkaline (pH 8-9) than in acidic (pH 5-6) water. In alkaline water cypermethrin, deltamethrin and fenvalerate have RC values approximately half those in acidic water. Permethrin is little affected by pH. Hydrolysis accounts for most of the degradation in high pH aquatic systems. Persistence results are based on an assumption of equivalent microbial degradation rates for these compounds with biolysis accounting for approximately 18% of the total breakdown. If biolysis is more important, the RC would be lower; conversely, if the contribution of microbes to degradation is minimal, the RC would be considerably longer. In general, the retentive capacities estimated for the synthetic pyrethroids are in excess of those for the other insecticides included in the relative hazard analysis (see Table 6.17). There are few reported field experiments with the synthetic pyrethroids which contain data appropriate to check model predictions. Rawn et al. (1982) examined the fate of permethrin (15 μg L^{-1}) in artificial outdoor ponds. Their data, produced by radio tracer techniques, indicated a rapid loss of permethrin from the water (half-life ~8.5 h) with corresponding increases in residue levels in the hydrosoil and duckweed. At 28 days post-exposure, concentrations of permethrin in the water were <0.01 μg L^{-1} while concentrations in hydrosoil and duckweed were 14 μg kg^{-1} and 0.04 μg g^{-1}, respectively. Permethrin was detectable (0.3 μg kg^{-1}) in the hydrosoil at 420 days but was not detectable elsewhere. Photolysis, was suggested as the mechanism of rapid degradation within the water column.

The PERSISTENCE MODEL predicts volatilization, rather than photolysis, will be the major pathway for permethrin removal. However, the predicted distribution, favoring accumulation in the benthic sediments (97%), is in general agreement with the data from the Rawn et al. (1982) study and studies of aerial applications of permethrin to forest ponds indicate the same general pattern (Kingsbury & Kreutzweiser 1979). In the latter example, detectable amounts of permethrin disappeared relatively rapidly from treated pond water (<0.25 μg L^{-1} within 48 hours

Table 6.16. Computer model predicted distribution and degradation of selected synthetic pyrethroid insecticides in a simulated shallow pond.

	Cypermethrin		Deltamethrin		Fenvalerate		Permethrin	
	pH 6	pH 8	pH 6	pH 8	pH 6	pH 8	pH 5	pH 9
Retentive capacity (RC) (days):	3987	1606	17058	9913	1328	853	3520	3175
First order t½ (days):	2763	1113	11819	6871	920	591	2440	2200
Percent of total load sequestered in:								
- water	4.7	4.7	0.5	0.5	5.2	5.2	0.8	0.8
- suspended solids	2.4	2.4	2.5	2.5	2.3	2.3	2.5	2.5
- benthic sediments	93.0	93.0	97.0	97.0	92.5	92.5	96.7	96.7
Percent removal due to:								
- volatilization	18.5	7.5	50.9	29.6	76.8	49.3	81.8	73.8
- photolysis	0.1	0.2	0.1	0.1	0	0	0	0
- hydrolysis	63.2	74.1	30.8	52.2	5.0	32.5	0	8.0
- biolysis[a]	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2

[a] The percent of total degradation that is microbial was arbitrarily chosen and input into the model.

Table 6.17. Computer model predicted distribution and degradation of selected major use agricultural insecticides in a simulated shallow pond.

	azinphos methyl		carbaryl		carbofuran		chlorpyrifos		diazinon		dimethoate		malathion		methoxychlor		phosmet	
	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9	pH 5	pH 9
Retentive capacity (RC) (days):	212	96	8	0.2	2444	0.7	678	486	14	16	222	4	842	0.2	298	297	6	0.5
First order t½ (days) (days):	147	66	6	0.1	1694	0.5	471	337	9	11	153	3	584	0.1	206	206	4	0.3
Percent of total load sequestered in:																		
- water	7.7	7.7	88.0	88.0	97.3	97.3	2.1	2.1	51.0	51.0	73.8	73.8	64.9	64.9	2.9	2.9	68.1	68.1
- suspended solids	2.3	2.3	0.3	0.3	0.1	0.1	2.4	2.4	1.2	1.2	0.6	0.6	0.9	0.9	2.4	2.4	0.8	0.8
- benthic sediments	90.0	90.0	11.7	11.7	2.6	2.6	95.4	95.4	47.8	47.8	25.6	25.6	34.2	34.2	94.6	94.6	31.1	31.1
Percent removal due to:																		
- volatilization	81.8	36.9	77.6	1.9	1.0	0	51.4	36.8	66.7	77.8	0	0	45.8	0	80.1	79.8	45.3	3.6
- photolysis[a]	0	0	3.9	0.1	0	0	0	0	0	0	0	0	0	0	0.1	0.1	0	0
- hydrolysis	0	45.0	0.3	79.8	80.8	81.8	30.5	45.0	15.2	4.0	81.8	81.8	36.0	81.8	1.7	1.9	36.5	78.2
- biolysis[b]	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2	18.2

[a] If quantum yield and extinction coefficient data are not available, photolysis is assumed to be unimportant and percent removal set equal to 0.

[b] The percent of total degradation that is microbial was arbitrarily chosen and input into the model

after application) and low levels of permethrin remained in benthic sediments at the conclusion of the study (0.01 ppm at 28 days). From limnocorral studies with permethrin, Solomon et al. (1985) found that the time to 90% dissipation in water was relatively short (13 to 27 days). Accumulation in the sediments was rapid and subsequent removal was slow over the 34-day observation period. The authors suggested that dissipation follows a biphasic reaction: a rapid phase of biological and chemical degradation occurring simultaneously with adsorption to sediments followed by a slower phase of sediment desorption and coupled degradation.

Studies of the fate of deltamethrin in small artificial ponds (Rawn et al. 1983) suggest that the distribution and primary loss vector are being correctly predicted by the computer model, however, the extremely long half-life and retentive capacity predicted for deltamethrin cannot be verified with existing field data. Rawn et al. (1983) found that the half-life of deltamethrin in the treated pond water was 2-4 h. Levels were less than 0.02 μg L^{-1} at 28 days but were detectable on days 35, 49 and 56, possibly due to releases from the breakdown of duckweed and algae. The undefined portion in suspended solids. Residues in the sediments were detected 112 days post-treatment and Rawn et al. (1983) suggested that these residues would persist over winter, indicating that the retentive capacity may exceed one year. Volatilization contributed 10-30% of the total loss during the initial four days after application. The computer model predicts that volatilization will account for 30-50% of deltamethrin loss.

A study of the distribution of cypermethrin in a natural pond demonstrated fate patterns similar to those of other synthetic pyrethroids. Concentrations in the water were highest within the first 24 h (1-1.4 μg L^{-1}) and were detectable (<0.01μg L^{-1}) for up to five weeks. Levels in the benthic sediments did not peak (6.9 μg kg^{-1}) until the fourth week after treatment at an application rate of 100 g ha^{-1} (Crossland 1982). One problem suggested by the Crossland study, as well as by a number of other field studies of pesticide persistence, was the difficulty in accounting for the mass-balance of insecticide after treatment. In this case, one hour after application the total mass of cypermethrin estimated in all compartments was equivalent to only 30-50% of the applied insecticide (Crossland & Stephenson 1979).

The computer model predicted distribution and degradation of selected carbamate, organophosphorous and organochlorine insecticides are summarized in Table 6.17. Carbaryl, carbofuran, dimethoate, malathion and phosmet remain primarily in water while azinphos methyl, chlorpyrifos and methoxychlor are mainly sequestered in benthic sediments. Diazinon is unique in that it partitions nearly equally between the water and sediments.

The persistence (RC) of azinphos methyl, carbofuran, chlorpyrifos, dimethoate, malathion and methoxychlor is greater than 150 days in an acidic pond. Carbofuran is the only compound with a retentive capacity and system half-life in the same order of magnitude as those of the synthetic pyrethroids under acidic conditions. Only methoxychlor, chlorpyrifos and azinphos methyl remain relatively persistent in alkaline water, although chlorpyrifos and azinphos methyl hydrolyzed somewhat more rapidly in the alkaline than in the acidic pond simulation. In contrast, carbofuran, dimethoate and malathion are greatly affected by system pH: they rapidly hydrolyze under basic pH conditions and disappear from the pond quickly (RC< 1 day). Carbaryl, diazinon and phosmet remain in the water column and are rapidly lost via volatilization and/or degraded through hydrolysis. These insecticides generally have RC values of less than 10 days in either acidic or alkaline ponds. The pH range of pothole ponds is wide, encompassing both acidic and alkaline systems but, in general, prairie potholes tend to be slightly alkaline (see Section 2.2).

Of those insecticides which the model predicts to be non-persistent, carbaryl's fate in natural pond ecosystems has been best studied. A review of early studies (Kuhr & Dorough 1976) indicated that carbaryl is to be non-persistent in natural waters. In addition, laboratory tests have confirmed the complete disappearance of carbaryl in water samples from various pond and marsh sources within one to two months (Szeto et al. 1979, Sharom et al. 1980b, Odeyemi 1982). However, recent applications of 840 g AI ha^{-1} to a natural pond showed that under certain circumstances this insecticide may remain at detectable levels in the aquatic environment beyond one year following application (Gibbs et al. 1984). In a pond with residues of 254 μg L^{-1} in water immediately after spray, detectable levels of 0.06 μg L^{-1} were found for up to 426 days. The test pond was acidic and the method of analysis did not distinguish between carbaryl and its major degradation product, 1-naphthol. Residue distribution in the sediments was patchy but levels were generally greater than 6 μg kg^{-1} for a period longer than 800 days. In a similarly treated pond with a lower initial concentration (5 μg L^{-1}), detectable water and sediment residues were found for less for less than 100 days. The results of additional tests suggested that the 100-day limit of detection within the system was probably the more accurate estimate of retentive capacity. The slope of the dissipation curves for carbaryl in all test ponds indicated an initial water and sediment half-life of less than one day. Although this field study showed an initially rapid breakdown of carbaryl as predicted by the computer model, the low level persistence of carbaryl was 10 times longer than suggested in the simulation. An unknown proportion of the persistence may be attributed to the experimenters inability to separate carbaryl and 1-naphthol.
The persistence of carbofuran, the other carbamate insecticide evaluated, also appears to be low, particularly under alkaline conditions (NRCC 1979). Charnetski et al. (1977) reported that an alkaline pond sprayed with carbofuran at 0.14 kg AI ha^{-1} had an initial water concentration of 4.3 μg L^{-1} which dropped to approximately 0.3 μg L^{-1} within eight days. Klaassen and eight days. Klaassen and Kadoum (1979) found carbofuran residues of 15 μg L^{-1} in water and 26-46 μg L^{-1} in mud in a farm pond 3 days after treatment with the with the insecticide at 25 μg L^{-1}; residues in surface water and mud were below 0.4μg L^{-1} by day 25 post-treatment.

Laboratory tests show that diazinon, like the carbamates, is degraded quickly in water (Sharom et al. 1980b). Diazinon persisted in rice paddy water for only five days after treatment and disappeared after 15 days from flooded soil that had been previously treated with the insecticide (Sethunathan 1972). In a microcosm fate study, Dortland (1980) reported a half-life for diazinon in water of 10 days. Persistence data for diazinon as a spray is unavailable for controlled field studies in pond or slough ecosystems. Hirano (1981) reported that the persistence in water of a paddy field following a granular application was less than 40 days and the initial half-life was less than two days. Persistence in salt marsh mud following a granular application of 227 g ha^{-1} was approximately 10 days (Marganian & Wall 1972).

The fate of azinphos methyl, chlorpyrifos and malathion has been somewhat more rigorously tested in laboratory microcosm, artificial dugout and natural pond studies. Azinphos methyl in simple water-sediment systems in laboratory conditions had a short half-life (4.7-5.1 days in water following a 1 mg L^{-1} treatment) under neutral to basic pH conditions (Dortland 1980). Outdoor tank studies by de Heer (1979) indicated that azinphos methyl was rapidly adsorbed to sediments as predicted by the NRCC PERSISTENCE MODEL. His field studies in irrigation ditches showed that the insecticide disappeared from water quite rapidly (half-life <4 days) with residue levels approaching the pre-treatment concentration in approximately 30 days. Unfortunately, there are no data indicating persistence within the sediments of these treated systems. Azinphos methyl had an average half-life in soil of about three months (The Chemagro Division Research Staff 1974), a value between the system half-life predicted for acidic and basic conditions in the pond simulation where azinphos methyl was primarily sequestered in sediments.

The results of a number of field assessments of chlorpyrifos persistence were summarized by NRCC (1978a). They suggested that the large losses soon after treatment are associated primarily with volatilization at the air-water interface and partitioning into various sorptive pools. These observations are in agreement with computer model predictions of greater than 50% removal via volatilization with greater than 98% of the remaining chlorpyrifos adsorbed to suspended solids and benthic sediments. In artificial pond studies, apparent half-life values in water in artificial pond studies ranged from approximately 1-3 days at application rates of 28-56 g ha^{-1} (Hurlbert et al. 1970, Macek et al. 1972, Rawn et al. 1978), to 0.1 days for an initial measured concentration of 20 μg L^{-1} (Hughes et al. 1980). Hughes et al. (1980) suggest that partitioning may occur more rapidly in a natural pond than in artificial dugouts. The time for 90% of applied chlorpyrifos to disappear from the aqueous phase was 1.6 days in a natural pond and 2.3-10 days in artificial ponds.

Dispersal to the sediments and persistence of chlorpyrifos in sorptive compartments is not well defined. In the case of the Hurlbert et al. (1970) data, equilibrium between water and sediments was established by about the seventh day post-treatment with an estimated concentration of 0.35 mg kg^{-1} in the sediments (NRCC 1978a). Subsequent clearance of chlorpyrifos from the system is thought to be slow. At an application rate of 1.12 kg AI ha^{-1}, Hurlbert et al. (1970) reported 0.14-0.52 mg kg^{-1} at day 14 post-treatment in the sediments. Marganian and Wall (1972) reported detectable levels of chlorpyrifos in the mud of an oversprayed salt marsh for more than 15 days. Due to the lack of both residue data for sediments and data for long-term low-level concentrations of this insecticide in water, there is little basis for further comparisons of field and computer-predicted persistence.

The few studies available describing malathion fate in artificial and natural aquatic ecosystems show that its life in water is short.

Dortland (1980) reported half-life values in water of 1.4-1.7 days in microcosms at pH 7.4-7.7 treated initially at 1 mg L^{-1}. Conte and Parker (1975) found residue levels in marsh water treated at 256 g ha^{-1} to be 0.8-3.2 μg L^{-1} two days after aerial application. In a similar study in a salt marsh, Tagatz et al. (1974) reported that, after three bi-weekly ULV aerosol sprays at 57 g ha^{-1}, a maximum aqueous residue of 0.49 μg L^{-1} was detected. In the latter study, the half-life of malathion in marsh water was <1 day. Residues in benthic sediments were undetectable (below 0.05 ppm) six hours after application but were recorded (0.28 ppm) in the cattail (Juncus) three days post-treatment. Residue data was unavailable from the replicated field study of malathion toxicity in artificial ponds (Kennedy & Walsh 1970). Although evidence is scant, persistence of malathion in slough-like ecosystems, as predicted by the computer, is probably short, particularly if the pH is in the basic range encouraging rapid hydrolysis.

Methoxychlor, the only organochlorine included in the hazard analysis, appears to persist for relatively long periods. Bradshaw et al. (1972) found methoxychlor residues of up to 5.2 μg L^{-1} in late spring and 1.2 μg L^{-1} in late fall in water samples from the Utah Lake drainage. Wallner et al. (1969) found that methoxychlor persisted for approximately six months at or above the 1 μg kg^{-1} level in bottom silt of a forest spring oversprayed in the control of European elm beetle (Scolytos multistraitus). Data on residues in Saskatchewan River water samples demonstrated that methoxychlor preferentially partitioned into suspended solids (>50%). Benthic sediment samples taken 22 km downstream from the input point accumulated from 0.05 to .1 mg kg^{-1} methoxychlor nine days after a 15 minute injection of 0.309 ppm of the insecticide (Fredeen et al. 1975). These data suggest that the computer-predicted distribution of this organochlorine, with approximately 95% in sediment, may accurately reflect the distribution in natural aquatic ecosystems. The few available data indicate relatively long retention of methoxychlor in sediments, however, there are no studies to substantiate the predicted system half-life of 200 days.

Persistence data for dimethoate is very limited and no microcosm or field studies are available describing the fate of phosmet. De Heer (1979) reported that dimethoate was not rapidly adsorbed to sediments and had a half-life of 3.6-13 days in sprayed irrigation ditches; the half-life in acidic water in outdoor tanks was much longer (half-life >70 days). The PERSISTENCE MODEL predicted a half-life of 153 days for dimethoate in the simulated shallow pond under acidic conditions (pH = 5).

6.3.8.2 Predicted hazard to aquatic invertebrates

In relative terms, the larger the hazard index value the greater the potential of hazard. However, until appropriate data on benchmark compounds are developed in field studies (see Section 6.3.10), the absolute value of calculated risk has little meaning other than providing the evaluator with general guidance in grouping compounds of high, moderate and low potential hazard, and in ranking the relative risk of compounds to invertebrates. Arbitrarily, for the purpose of the present discussion, index values >100 are designated high hazard, those in the range of 10-100 are of moderate hazard, and those <10 are of low hazard.

During the initial phase of exposure, the predicted hazard to aquatic invertebrates is high for compounds which are highly toxic and readily available in the aqueous phase. Availability is primarily determined by insecticide input (AR) and volatilization and hydrolysis losses. The relative hazard index (RHI) values in Table 6.18 show general consistency in ranking high and low risk insecticides. The synthetic pyrethroid compounds fall primarily into the high hazard category along with azinphos methyl, chlorpyrifos, malathion and methoxychlor, each having an RHI value> 100 for two or more taxa. There are, however, also some notable taxonomic differences in sensitivity and some critical data gaps, most obviously for carbofuran, phosmet and deltamethrin.

Acute hazard in an acidic pond, based on Chironomus, is high for cypermethrin, chlorpyrifos and malathion and moderate for the rest of the compounds for which data were available. The pattern was much the same for the mosquito genus, Culex. The data bases for both taxa, however, are incomplete. Of notable absence are data for azinphos methyl and dimethoate, compounds of high and low risk, respectively, according to model calculations based on other invertebrate taxa.

In contrast, under pond conditions simulating an acidic environment, acute hazard defined by Daphnia toxicity is relatively high for fenvalerate and permethrin but appears low to negligible for cypermethrin. The low estimate of potential hazard for cypermethrin can be attributed to its 24 h LC_{50} value of 2 $\mu g\ L^{-1}$ which is approximately a factor of 10 higher than median lethal values for fenvalerate and permethrin. Risk calculations based on daphnid toxicity were not made for deltamethrin as the 24 and 48 h LC_{50} values of 5 and 8 $\mu g\ L^{-1}$ (Hoechst Canada Inc., unpublished) appear to be inconsistent with most 24-96 h LC_{50} values for other aquatic invertebrate species which are less than 1 $\mu g\ L^{-1}$ (see Tables 6.2-6.6). Predicted hazard under acidic conditions is high for simulated exposures of azinphos methyl, malathion and methoxychlor. Moderate hazard is predicted for carbaryl, chlorpyrifos, diazinon and phosmet with little risk of community damage expected from dimethoate exposures at recommended application rates.

The ACUTE EXPOSURE MODEL suggests, based on Gammarus spp. which is the most sensitive species, that cypermethrin and fenvalerate present a high relative risk. The potential hazard from deltamethrin and permethrin cannot be assessed due to the unavailability of appropriate toxicity data. In comparison with the predictions based on Daphnia, chlorpyrifos and phosmet should be reclassified in the high hazard category with diazinon shifted to the lowest risk classification. Estimated hazard is generally in close agreement with that calculated for the two dipteran taxa.

Based on the stonefly and mayfly toxicity (Pteronarcys and various mayfly species), permethrin and deltamethrin present the greatest ecological hazard followed by that posed by methoxychlor exposure. The relative hazard index values estimated are, in general, lower for these insects than for the other invertebrates chosen. A majority of the compounds would be ranked as presenting a moderate risk. Diazinon and dimethoate would present a low potential hazard as previously suggested from gammarid toxicity data.

Table 6.18. Relative risk index values[a] for *Daphnia magna*, *Gammarus lacustris*, Aquatic Insects, *Chironomus* spp., *Culex* spp. or comparable taxa, based on the ACUTE EXPOSURE MODEL.

	ACUTE EXPOSURE MODEL									
	Pond pH 5					Pond pH 9				
Insecticide	Daphnia	Gammarus	Aquatic Insects	Chironomus	Culex	Daphnia	Gammarus	Aquatic Insects	Chironomus	Culex
Synthetic Pyrethroids										
cypermethrin	8	128-278p	28Cd	53-160	229	8	124-270p	27Cd	53-160	229
deltamethrin	NA[b]	NA	217Bp	21-26	33-304	NA	NA	212Bp	21-25	32-297
fenvalerate	8	400-467ps	15-78E	4-13	12-14	8	388-453ps	15-76E	4-13	12-14
permethrin	62-187	NA	682Br	NA	75-165	62-182	NA	673Br	NA	73-100
Other Insecticides										
azinphos methyl	78-109	415-446	16-42Pc	NA	NA	52-72	275-295	10-28Pc	NA	NA
carbaryl	31	10-11	14-23Pc	22	< 1	2	< 1	< 1Pc	1	< 1
carbofuran	4	NA	NA	NA	NA	< 1	NA	NA	NA	NA
chlorpyrifos	17	331-606	5-7Pc	227-544	170-227	15	307-558	5-7Pc	200-480	150-200
diazinon	70-84	< 1	1-2Pc	NA	8-11	75-89	< 1	1-2Pc	NA	9-11
dimethoate	< 1	< 1	< 1-2Pc	NA	NA	< 1	< 1	< 1Pc	NA	NA
malathion	254-564	127-141	13-15Pc	152-378	33-35	8-17	4-5	< 1Pc	5-11	1
methoxychlor	169	133-240	21-112Pc	96	34-71	169	133-240	21-112Pc	96	34-71
phosmet	29	428f	NA	NA	NA	2	26f	NA	NA	NA

---

[a] Relative risk index values were calculated from a single data set, or from the high and low median lethal exposure values if a range of data was available. The index was calculated from toxicity data for *Gammarus fasciatus* (f), *Gammarus pulex* (p), *Gammarus pseudolimnaeus* (ps), *Cloeon dipterum* (Cd), *Baetis parvus* (Bp), *Baetis rhodani* (Br), *Ephemerella* sp. (E), *Pteronarcys californica* (Pc)

[b] NA - no appropriate data set available

Data for the alkaline pond (pH 9) show that rapid hydrolysis under high pH conditions reduces the short-term risk of carbaryl and malathion to negligible levels. The synthetic pyrethroids, azinphos methyl, chlorpyrifos and methoxychlor are consistently in moderate to high risk positions within the rankings. The lack of toxicity data for carbofuran limits confidence in model rankings. This is unfortunate as carbofuran is widely used for insect control in prairie agriculture.

A listing of the weighted average ranking of potential short-term hazard to aquatic invertebrates based on calculated RHI values is presented in Table 6.19. The acidic and alkaline ponds show very similar rankings of compounds. With the exception of fenvalerate, the pyrethroids are in the top half of the risk lists along with azinphos methyl, chlorpyrifos and methoxychlor. These are followed by phosmet, malathion and fenvalerate. Carbaryl, carbofuran and dimethoate are at the bottom of the list under both scenarios. The rankings of malathion, azinphos methyl and diazinon are affected by pond pH, with the first two showing decreased risk with increasing pH and the latter posing a greater risk at higher pH. The value of the model in ranking relative hazard is most apparent for those compounds which are substantially affected by environmental conditions (eg. pH).

The range of relative hazard values calculated under the CHRONIC EXPOSURE MODEL is substantially greater than that predicted by the initial phase analysis (Table 6.20). Compounds presenting high potential hazard have index values from several hundred to several thousand. Malathion, under acidic conditions, displays the largest calculated value of relative hazard. The influence of hydrolysis in alkaline waters is again reflected in the index values for most compounds, particularly that of malathion which decreases in value by three to five orders of magnitude in an alkaline pond.

Table 6.21 lists the selected insecticides by relative hazard rankings based on the retentive capacity and the weighted average rank of RHI values calculated with the CHRONIC EXPOSURE MODEL. The influence of long retentive capacity on risk calculations is seen in the grouping of the synthetic pyrethroid compounds at the top portion of all lists. Under acidic conditions, malathion is ranked as presenting the greatest potential risk due to its tendency to partition into water and its slow hydrolytic breakdown. Carbofuran, likewise, has a high ranking which is opposite that predicted by the ACUTE EXPOSURE MODEL, but its ranking is based on a single data set. The low RC of malathion and carbofuran under basic pH conditions is apparent in their large drop in relative rankings. At pH 9, carbofuran and malathion are predicted to be of negligible hazard. Azinphos methyl, chlorpyrifos and especially methoxychlor posed a much greater risk at higher pH. The long-term ecological hazard to invertebrates from carbaryl, diazinon and dimethoate is predicted to be low under acidic conditions and diminished further under alkaline conditions.

6.3.9 Empirical analysis of the influence of physico-chemical data on predicted hazard

Table 6.19. Lists of insecticides reflecting relative hazard rankings based on ACUTE EXPOSURE MODEL predictions and short-term laboratory toxicity test data[a].

Toxicity Test Data	ACUTE EXPOSURE RISK MODEL		
	Acidic Pond	Alkaline Pond	
deltamethrin	permethrin	permethrin	high risk
cypermethrin	azinphos methyl	chlorpyrifos	
fenvalerate	chlorpyrifos	deltamethrin	
permethrin	deltamethrin	methoxychlor	
azinphos methyl	methoxychlor	cypermethrin	
malathion	cypermethrin	azinphos methyl	
methoxychlor	phosmet	fenvalerate	
chlorpyrifos[b]	malathion	diazinon	
carbaryl[b]	fenvalerate	phosmet	
diazinon	carbaryl	malathion	
phosmet	diazinon	carbaryl	
carbofuran[c]	carbofuran[c]	dimethoate	
dimethoate	dimethoate	carbofuran[c]	low risk

[a] Toxicity ranking based on average of 24 and 48 h LC_{50} values for five taxa; model rankings are a weighted average of ranking for individual taxa accounting for the number of compounds for which toxicity data was available (the weighting factor is the ratio of the total number of compounds over the number of compounds for which we have an RHI value)

[b] Of equal rank within individual list

[c] Ranking based solely on toxicity data for Daphnia

Table 6.20. Relative risk index values[a] for Daphnia magna, Gammarus lacustris, Chironomus spp., Culex spp., Aquatic Insects, or comparable taxa, based on the CHRONIC EXPOSURE MODEL.

	CHRONIC EXPOSURE MODEL									
	Pond pH 5					Pond pH 9				
Insecticide	Daphnia	Gammarus	Aquatic Insects	Chironomus	Culex	Daphnia	Gammarus	Aquatic Insects	Chironomus	Culex
Synthetic Pyrethroids										
cypermethrin	323	4964-10755p	1076Cd	2154-6460	9238	130	1999-4331p	433Cd	867-2600	3718
deltamethrin	NA[b]	NA	3932Bp	381-472	590-5505	NA	NA	2286Bp	222-274	343-3200
fenvalerate	125	5912-6898ps	224-1150E	55-193	177-209	81	3797-4430ps	144-738E	35-124	113-133
permethrin	404-1212	NA	4406Br	NA	485-1067	364-1093	NA	3975Br	NA	437-962
Other Insecticides										
azinphos methyl	264-367	1408-1508	53-141Pc	NA	NA	119-165	634-679	24-63Pc	NA	NA
carbaryl	69	22-24	32-51Pc	48	2-3	2	< 1	< Pc	1	< 1
carbofuran	2047	NA	NA	NA	NA	< 1	NA	NA	NA	NA
chlorpyrifos	61	1226-2229	20-27Pc	811-1952	610-811	44	879-1597	14-20Pc	585-1408	440-585
diazinon	155-187	< 1	3-5Pc	NA	18-22	183-218	< 1	4-5Pc	NA	21-27
dimethoate	< 1-2	11-12	41-57Pc	NA	NA	< 1	< 1	< 1Pc	NA	NA
malathion	28222-62714	14114-15679	1411-1613Pc	16932-42018	3700-3920	6-14	3-4	< 1Pc	4-9	< 1
methoxychlor	24	19-34	3-16Pc	14	5-10	430	339-611	53-284Pc	245	86-181
phosmet	47	702f	NA	NA	NA	2	22f	NA	NA	NA

[a] Relative risk index values were calculated from a single toxicity data set, or from the high and low median lethal exposure values if a range of data was available. The index was calculated from toxicity data for Gammarus fasciatus (f), Gammarus pulex (p), Gammarus pseudolimnaeus (ps), Cloeon dipterum (Cd), Baetis parvus (Bp), Baetis rhodani (Br), Ephemerella sp. (E), Pteronarcys californica (Pc).

[b] NA - No appropriate data set available

Table 6.21 Lists of insecticides reflecting relative hazard rankings[a] based on CHRONIC EXPOSURE MODEL calculations and retentive capacity predictions.

Retentive Capacity		CHRONIC EXPOSURE MODEL		
Acidic Pond	Alkaline Pond	Acidic Pond	Alkaline Pond	
deltamethrin	deltamethrin	malathion	permethrin	↑ high risk
cypermethrin	permethrin	carbofuran[b]	cypermethrin	
permethrin	cypermethrin	cypermethrin	deltamethrin	
carbofuran[b]	fenvalerate	permethrin	azinphos methyl	
fenvalerate	chlorpyrifos	deltamethrin	chlorpyrifos	
malathion	methoxychlor	azinphos methyl	methoxychlor	
chlorpyrifos	azinphos methyl	fenvalerate	fenvalerate	
methoxychlor	diazinon	chlorpyrifos	diazinon	
dimethoate	dimethoate	phosmet	phosmet	
azinphos methyl	carbofuran[b]	carbaryl	malathion	
diazinon	phosmet	diazinon	carbaryl	
carbaryl	malathion	dimethoate	dimethoate	
phosmet	carbaryl	methoxychlor	carbofuran[b]	↓ low risk

[a] Model ranking is weighted average of rankings for individual taxa accounting for the number of compounds for which toxicity data was available (ratio of the total number of compounds for which we have an RHI value)

[b] Ranking based solely on toxicity data for Daphnia

Because the available literature often provides a range of potential input data for persistence simulations (see Table 6.12), an empirical analysis of the influence of extreme values on predicted risk was undertaken. The breadth of the range of values for solubility and octanol-water partition coefficients for the synthetic pyrethroids is particularly wide and could substantially weaken confidence in persistence predictions in those cases where there is no generally accepted parameter value. Fortunately, the ACUTE EXPOSURE MODEL is robust because of its default calculation system and is therefore little affected by substantial changes in the value of most physico-chemical parameters. As expected, changes in input parameter values have a more dramatic effect on relative hazard index values predicted by the CHRONIC EXPOSURE MODEL. For example, a four-fold decrease in the solubility value input for cypermethrin (0.04 to 0.009 mg L^{-1}) did not influence relative hazard calculated by the short-term model but did reduce absolute index values by nearly half under conditions of equilibrated distribution. However, cypermethrin's ranking remained in the high hazard category. A 70-times increase in K_{OW} (29,500 to 2,100,000) again did not affect risk in the initial phase of insecticide dissipation but did increase the absolute value of the relative hazard index by approximately 30% according to the chronic model. Much the same situation occurred with solubility and K_{OW} changes in input data for fenvalerate. Use of the low values of solubility (0.002 mg L^{-1}) and K_{OW} (26,500) produced relative risk values less than half those predicted under the initial conditions. Under conditions of minimum persistence, fenvalerate was classified as borderline between the moderate and high hazard categories using the ACUTE EXPOSURE MODEL and between the low to moderate categories using the CHRONIC EXPOSURE MODEL. Using a high K_{OW} value (1.5×10^6) had little effect on short-term hazard but increased predicted long-term hazard. The pattern for permethrin was the same.

There are further examples of the models' robustness from analysis of the other selected insecticides. Tripling the solubility for carbaryl raised calculated hazard index values by <2 times and did not change its classification. A near doubling of the octanol-water partition coefficient for chlorpyrifos had a negligible impact on short-term risk estimates and approximately doubled index values in the long-term model calculation. Use of the larger vapour pressure value for malathion (1.25×10^{-4} mm Hg), a factor of 20 greater than the select value, had no impact on the ACUTE EXPOSURE MODEL calculations but cut the absolute value of the index in the CHRONIC EXPOSURE MODEL by half. The relative rankings of chlorpyrifos and malathion were not substantially affected by the changes in input values. A change of the K_{OW} for methoxychlor from 2,500 to 47,500 had little influence on the predicted hazard in either model.

One aspect of the NRCC PERSISTENCE MODEL which does affect risk estimates in the CHRONIC EXPOSURE MODEL calculations is the relationship between large K_{OW} values and predicted fractional distribution of the insecticide among aquatic compartments. At K_{OW} values $>3 \times 10^6$, an insecticide is predicted to be sequestered entirely within the suspended particulate and benthic sediment compartments. With no portion of the compound in the water ($f_1 = 0$), the calculated relative risk index equals zero for invertebrates exposed via water (see equation 2, Section 6.3.3). For example, in risk calculations for fenvalerate, use of the largest reported K_{OW} value (6.3×10^6) produced a RHI of zero, but use of the next lower value (1.5 x 10^6) produced risk predictions based on the five invertebrate taxa which were all greater than 1,000 and in the high hazard category. It should be noted that in such cases of complete sequestering of an insecticide into sediments, organisms which actively feed in the sediments may be at substantial risk even though their open-water counterparts are not likely to be exposed. In addition, the relative risk of hazard from acute exposure may be high.

Although hazard estimation is relatively insensitive to large changes in single physico-chemical parameters, index values are extremely sensitive to toxicity inputs. In this sense, the models are weighted towards toxicity information. The accuracy of LC_{50} data and the validity of the integrated median lethal toxicity model are critical to model predictions. This knowledge has spawned the multi-taxa approach (Daphnia, Gammarus, Pteronarcys, Chironomus, Culex) to the prediction of hazard to the invertebrate community. Further consideration should be given to weighting the sensitivity and ecological (trophic) importance of specific taxa in average rankings.

6.3.10 Benchmark calibration

Carbaryl and permethrin, which produce moderate and high levels of adverse impact on aquatic invertebrates, respectively, are possible benchmarks to be used to calibrate RHI values derived under single input acute exposure conditions so that they reflect the severity of impact expected under field conditions. Although neither compound has been previously considered in a benchmark capacity, both have been subjected to long-term controlled field studies which provided data relating exposure to the magnitude and duration of effects (Table 6.22). A moderately toxic insecticide such as carbaryl is expected to deplete invertebrate numbers to approximately 30-80% of normal levels and biomass to 60-75% of pretreatment standing crop. Certain species which are important to duckling feeding (e.g. chironomids) will remain relatively unaffected. The lag period before the re-establishment of normal numbers and biomass is expected to be 3-6 months. A high impact compound such as permethrin is expected to reduce invertebrate population abundance by >80%. Benthic macroinvertebrate biomass in treated systems may be reduced by >70%. Possibly only aquatic worms, molluscs and rotifers will be unaffected. Full recovery of community numbers and biomass is expected to be delayed for more than six months and may take longer than one year, depending on the availability of mobile colonizers.

Insecticides with RHI values for acute exposure conditions consistently greater than 100 (see Table 6.18) are expected to produce effects equivalent in severity to those summarized for permethrin. Cypermethrin, deltamethrin, permethrin, azinphos methyl and chlorpyrifos are associated with the high impact benchmark and malathion is predicted to induce similar effects under acidic conditions. Compounds with values between 10 and 100, such as methoxychlor, should produce moderate impacts similar to those described for carbaryl. Diazinon and dimethoate (RHI <10) present little or no risk with effects expressed primarily through community changes.

Table 6.22. Summary of expected adverse response of aquatic invertebrate community in treated[a] ponds for suggested benchmark insecticides. Carbaryl impacts are classified as moderate and permetherin impacts as severe.

Insecticide	Reductions in Aquatic Invertebrates[b] Pre-treatment (Control)	Post-treatment	Percent Reduction	Time Required to Recover	General Trends	Comments	References
carbaryl	benthic ~ 300-1,000 ind. m^{-2} ~ 1.5 g dry wt. m^{-2}	~ 200 ind. m^{-2} ~ 0.4 g dry wt. m^{-2}	~ 30-80 ~ 75	little effect on chironomids; 3 weeks for ephemeropterans; up to 1 year amphipods; 20 weeks for total numbers of benthic organisms; no data on recovery of open water community	elimination of sensitive populations (amphipods) for an extended period; chironomids become dominant; seasonal reductions in available macroinvertebrate biomass	ducklings on treated ponds weighed 30% less than those on untreated ponds and had only 40% the growth rate of non-food-stressed ducklings over the initial 10 days post-treatment associated with treatment	Gibbs *et al.* 1984 Hunter *et al.* 1984
	surface water ~ 60-180 ind. m^{-3} ~ 0.008 g m^{-3}	~ 20-30 ind. m^{-3} ~ 0.003 g m^{-3}	~ 70 ~ 60				
permethrin	benthic ~ 2,800 ind. m^{-2} ~ 1.4 g dry wt. m^{-2}	~ 200 ind. m^{-2} ~ 0.1 g dry wt. m^{-2}	> 90 > 90	oligochaetes and molluscs least affected; up to a year for many crustaceans and insects; 16 weeks for macrozooplankton; rotifers actually increased in numbers	elimination of most free-swimming and benthic arthropods; invertebrate population may remain depleted for a year or more	alterations of reproduction potential and transient effects on growth rates of fish in treated systems	Kingsbury 1976 Kingsbury and Kreutzweiser 1979
	open water ~ 300-1,500 ind. m^{-3}	~ 100-150 ind. m^{-3}	80				
	near macrophytes ~ 0.5-2.8 g dry wt. m^{-3}	< 0.3 g dry wt. m^{-3}	70				cf. NRCC 1985
	zooplankton ~ 100-200 ind L^{-1}	< 1 ind. L^{-1} (cladocerans, copepods)	99				Kaushik *et al.* 1985

---

[a] Treated at application rates approximately equivalent to those presently recommended for crop protection

[b] Estimated average reduction for the initial 30 days post-treatment; biomass if unreported was estimated from taxonomic composition and abundance data from authors and mean dry wt data from Driver et al. 1974.

There is an obvious need to further refine the benchmark data for macroinvertebrate impacts with the currently selected compounds and to develop benchmark data for a compound ranked intermediate between carbaryl and permethrin. Greater attention must be paid to sampling design to allow effects to be quantified for invertebrate communities associated with open water (also surface water), benthic and macrophyte habitats in sloughs. Relationships of standing crop biomass and individual size frequencies within the various habitats needs to be established. In addition, aquatic impact studies need to be expanded to include analysis of macroinvertebrate production to biomass relationships, particularly during the productive summer period. The refinement of current benchmarks and the addition of others will facilitate more accurate characterization of relative hazard values.

6.3.11 Critique of relative hazard analysis

The greatest value of a hazard analysis approach is the framework that it provides for comparing persistence and toxicity data. Data requirements are minimal, the NRCC PERSISTENCE MODEL is well developed and user-friendly and relative risk calculations are simple. Because the predicted loss and transfer coefficients can reflect the influence of environmental factors such as pH and temperature, relative hazard estimates provide more ecologically relevant data than rankings based on standard toxicity test results alone. Both the persistence and toxicological component of the ACUTE EXPOSURE MODEL have a sound theoretical base. The hazard analysis approach appears promising for screening insecticides that may, through elimination of invertebrates, adversely impact the trophic resource base of predators such as ducks and fish.

Specific advantages of the proposed relative hazard models include:

- the relative robustness and flexibility of the PERSISTENCE MODEL,
- the minimal data requirements provide the pesticide manufacturers with a clear picture of the evaluators' specific information needs for initial screening,
- the standard format facilitates the comparison of relative risk of insecticides to aquatic invertebrates,
- the potential for modification to examine impacts of insecticides on other species (eg. ducklings) is possible.

Problem areas include:

- the nonavailability of critical toxicity and aquatic fate data for a substantial number of insecticides,
- the present inability to adequately account for biolysis in persistence predictions,

- the present inability of the PERSISTENCE MODEL to account for macrophyte contributions to insecticide removal and degradation,
- the integrated median lethal exposure toxicity benchmark may not be the most appropriate form of toxicity data input for the CHRONIC EXPOSURE MODEL,
- uncertainty estimates are not attached to persistence or hazard estimates,
- the absolute value of the RHI is difficult to interpret without further development of adequate benchmark data from field evaluations.

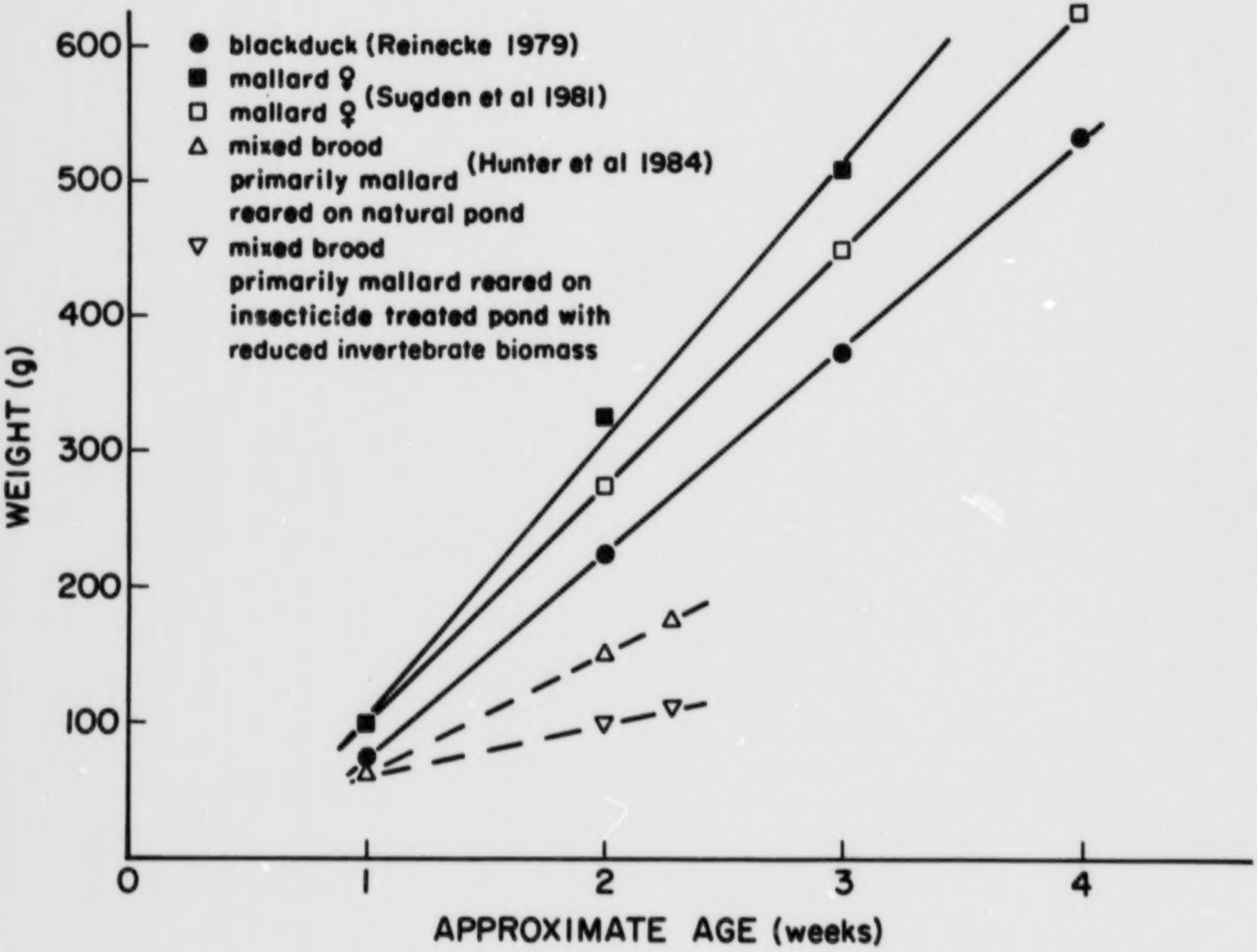
6.3.12 Hazard to ducklings

The data discussed in Section 2.5 established that duckling growth in natural habitats during the initial two weeks for most species is highly dependent on invertebrate protein. Dependency is much longer for species such as Lesser Scaup and Pintail (Sugden 1973, 1982). Therefore, a direct relationship between aquatic macroinvertebrate abundance (biomass) and duckling growth is expected during the initial 2-4 weeks. In addition, the chances of survival in slow-growing individuals are low because resistence to chilling increases in proportion to bird weight (Cross 1966) and plumage development (Sturkie 1954). Also, increased searching activity to compensate for low prey availability is energetically inefficient and may place the ducklings at an increased risk of predator attack while feeding (J.-L. Desgranges CWS Quebec Region, pers. comm.). Overland migrations in search of potholes with adequate food resources also make broods vulnerable to predation losses (see Section 2.5). Hence, fast growth in ducklings is ecologically beneficial and, conversely, slow- or no-growth places broods at high risk of mortality.

6.3.12.1 Experimental evidence

Carbaryl has been implicated as an indirect cause of reduced duckling growth because of its effect on the quality and quantity of duckling food resources (Hunter *et al*. 1984). Therefore, insecticides of equal or greater relative hazard are expected to adversely affect ducklings. The work by Hunter and colleagues (1984) is significant in that it was the first published study which looked at the link between aquatic system response to insecticide stress and the impact on waterfowl feeding in the stressed ecosystem. Figure 6.3 graphically presents growth data on Mallards (Sugden *et al*. 1981) and Black Ducks (Reinecke 1979) under optimal laboratory rearing conditions and from uncontaminated and contaminated pond habitats (Hunter et al. 1984). As expected, ducklings grew more rapidly under laboratory conditions with food (commercial poultry starter, 24% protein) supplied *ad libitum*. Under these optimal conditions, ducklings of both species gained, on average, 150 to 205 g week^{-1} from age one to four weeks. Weight gain was a function of age (body weight) and dietary intake.

Figure 6.3. Plot of growth for Black Duck and Mallard ducklings during weeks one through four of age. Optimal growth rates for laboratory research birds are compared to those for experimental broods reared in natural habitats under conditions of sufficient and insufficient invertebrate foods.


● blackduck (Reinecke 1979)
■ mallard ♀
□ mallard ♀
(Sugden et al 1981)
△ mixed brood
primarily mallard
reared on natural pond
(Hunter et al 1984)
▽ mixed brood
primarily mallard reared on
insecticide treated pond with
reduced invertebrate biomass
WEIGHT (g)
600
500
400
300
200
100
0
1
2
3
4
APPROXIMATE AGE (weeks)

A mixed brood (five Mallards and one Black Duck) feeding naturally on aquatic invertebrates in an uncontaminated reference pond grew at a rate of ~90 g $week^{-1}$ during the same growth period. Ducklings (five Mallards and two Black Ducks) reared on a carbaryl-treated pond, supporting a macroinvertebrate biomass approximately 75% of that in the reference pond, grew at a rate of ~40 g $week^{-1}$. The slopes of the cumulative plots suggest that ducklings raised under uncontaminated natural conditions may grow at 50-70% of their potential rate. This difference may be partially attributed to additional energy expenditures required for searching and catching prey under natural conditions as well as those for maintenance of body temperature. Experimental artifacts in the field such as shortened feeding periods and handling effects may also contribute to the less than optimal weight gain. There is some evidence that ducklings allowed to forage in seminatural conditions with little handling grow at rates near those measured in laboratory feeding studies (E. Driver CWS Prairie Migratory Bird Research Centre, pers. comm.). Cummulative growth curves for wild and laboratory reared Lesser Scaup were nearly identical for the initial five weeks (Sugden & Harris 1972).

Food-stressed ducklings in the carbaryl-treated pond grew at only half the rate of the matched brood on the reference pond and at 30% of their optimal rate. Recent evidence suggests that, under conditions of more massive resource depletions, ducklings may die. On experimentally acidified artificial ponds, 50% of Black ducklings lost weight and 25% died; only 25% were able to maintain their weight over a 10-day experimental period (M. Haramis U.S.F.W.S. Patuxent and Maryland, pers. comm.). In contrast, 85% of the control brood gained weight and the remainder maintained their weight. Black ducklings on natural lakes with a large fish population and a low abundance of open water macroinvertebrates also grew less rapidly than experimental broods on more productive lakes without fish (J.-L. Desgranges CWS Quebec Region, pers. comm.). In all of these experimental studies there was documentation of substantially increased searching-moving activity in stressed ducklings and of loss of cohesiveness within the broods.

Although the direct experimental evidence is minimal, the ecological concept of carrying capacity strongly suggests a relationship between low invertebrate standing crop (resulting from chemical stress) and reduced duckling growth (and survival) which may allow crude predictions of secondary trophic impacts based on RHI values. As a first approximation, the surface sweep net estimates of invertebrate biomass from Hunter et al. (1984) were regressed against duckling growth rate (see Table 6.22 and Figure 6.3). This approach is, at present, unsatisfactory for several reasons. The most obvious problem is the lack of sufficient data to compare growth with food supply at or below maintenance levels. Also problematic are the very low invertebrate biomass estimates which Hunter and colleagues associate with their uncontaminated test ponds. Their technique of sweeping the surface and to a depth of only 15 cm provided biomass estimates a factor of 10 lower than others reported for surface and water column samples (see Section 2.4). Differentiation between treatment and control biomass would be exceedingly difficult if the low estimates were correct considering inherent natural variability in macroinvertebrate community standing crop. Under such conditions, relating estimates of prey availability to growth is difficult. However, the value of developing such an empirical relationship is obvious and should be further explored experimentally with the appropriate considerations of sampling design. Studies designed to develop an empirical measure of duckling growth and survival with respect to prey availability are currently under way in artificial ponds at the Northern Prairie Wildlife Research Center, Jamestown, North Dakota (G. Swanson, pers. comm.) and through the Department of Natural Resources, Minnesota (R. Laurenz, pers. comm.).

6.3.12.2 Duckling growth models

Alternatively, the problem of estimating hazards to ducklings may be approached through energetic considerations. Bioenergetic requirements have been developed for some of the major species, for example Mallards (Sugden *et al*. 1981), Lesser Scaups (Sugden & Harris 1972, Lightbody & Ankney 1984) and Black Ducks (Penney & Bailey 1970, Reinecke 1979). These studies have established relationships between dietary intake, growth rate, live weight and age of ducklings for laboratory reared birds. There are some striking similarities among the growth curves for various species, although the slope of the growth plots can differ. Ducklings gain weight rapidly until about seven weeks of age. Increases in live weight are generally linear during the maximum growth period age of one to four weeks. At the age of 2-3 weeks the maximum rate of weight change is achieved. This is also the age at which the greatest quantities of invertebrates are eaten by many species (Sugden 1973). Food intake increases to a maximum in week four or five and declines thereafter. The efficiency of converting food to growth, expressed as the ratio of live weight change to dry matter intake decreases linearly with increasing age and ranges from a low of 0.35 for week four to a high of 0.60 for week two for the species mentioned above. Bioenergetic efficiencies do differ between species but differences are not large and appear to be related to live weight as well as to the age of the ducklings.

If the relationships between age, live weight, dry matter intake and optimal growth rate are known and relatively comparable for a number of species, a generalized growth rate can be calculated as a function of dietary intake as long as certain assumptions on the efficiency of converting dry weight of food into live weight of duckling are made. Estimates of necessary dietary intake to meet specified growth requirements can then be related to available invertebrate biomass using relationships on average time spent feeding, feeding success per unit time, consumption rate per prey density and energy-biomass conversions for macroinvertebrates.

The suggested DUCKLING GROWTH MODELS are based on growth, consumption and feeding success data for the Mallard as a representative dabbler and the Lesser Scaup as a representative diver with a long period of invertebrate dependency (Sugden 1971, Sugden & Harris 1972, Pehrsson 1979). Plots of live weight, dry matter intake and weight change for age one to four weeks are presented in Figure 6.4 for the Mallard and Figure 6.5 for the Lesser Scaup.

Figure 6.4. Plots of average cumulative growth, dry matter intake and live weight change for captive Mallard ducklings during their first four weeks (data from Sugden et al. 1981).

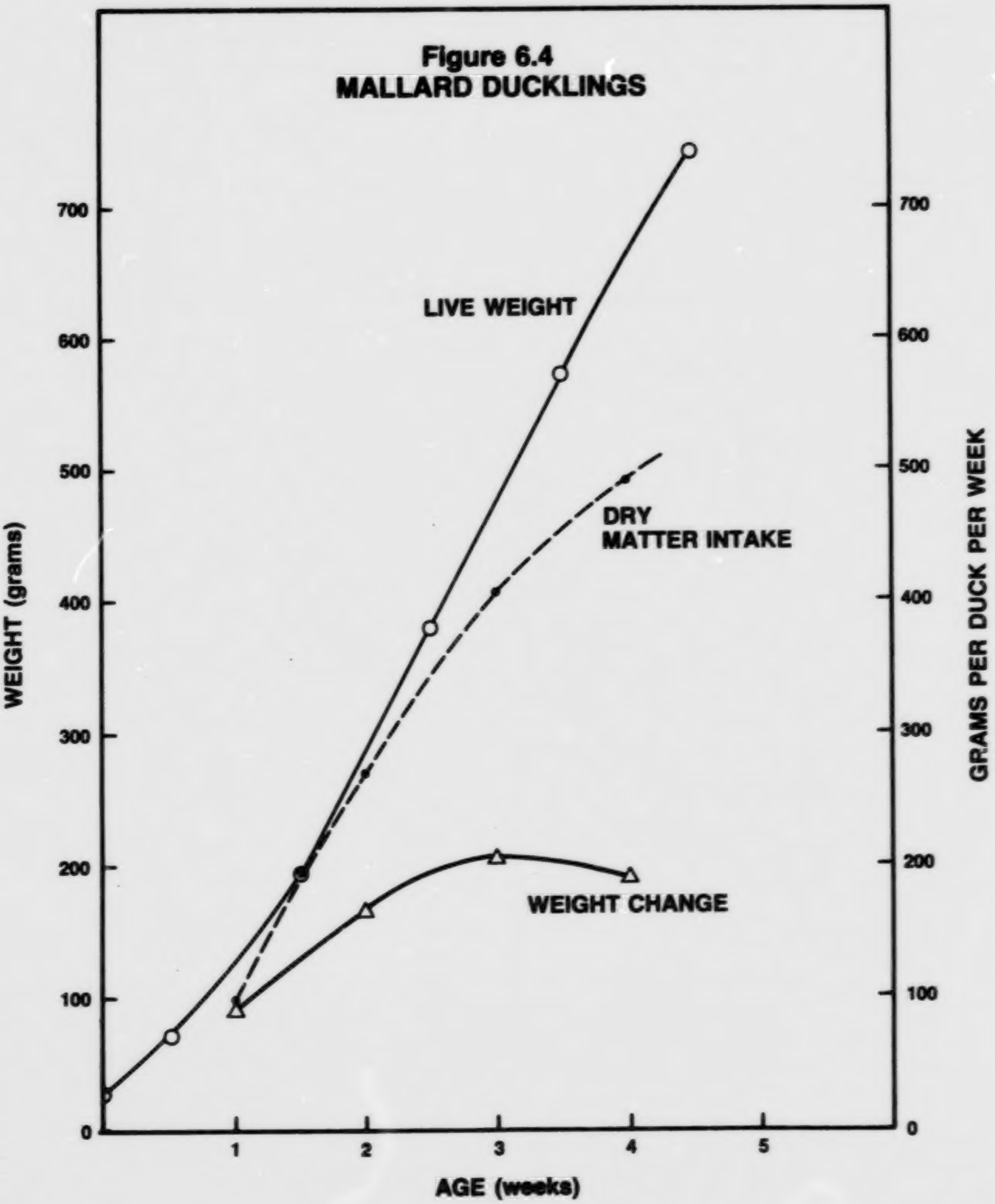


Figure 6.4
MALLARD DUCKLINGS

Figure 6.5. Plots of average cumulative growth, dry matter intake and live weight change for captive Lesser Scaup ducklings during their first four weeks (data from Sugden & Harris 1972).

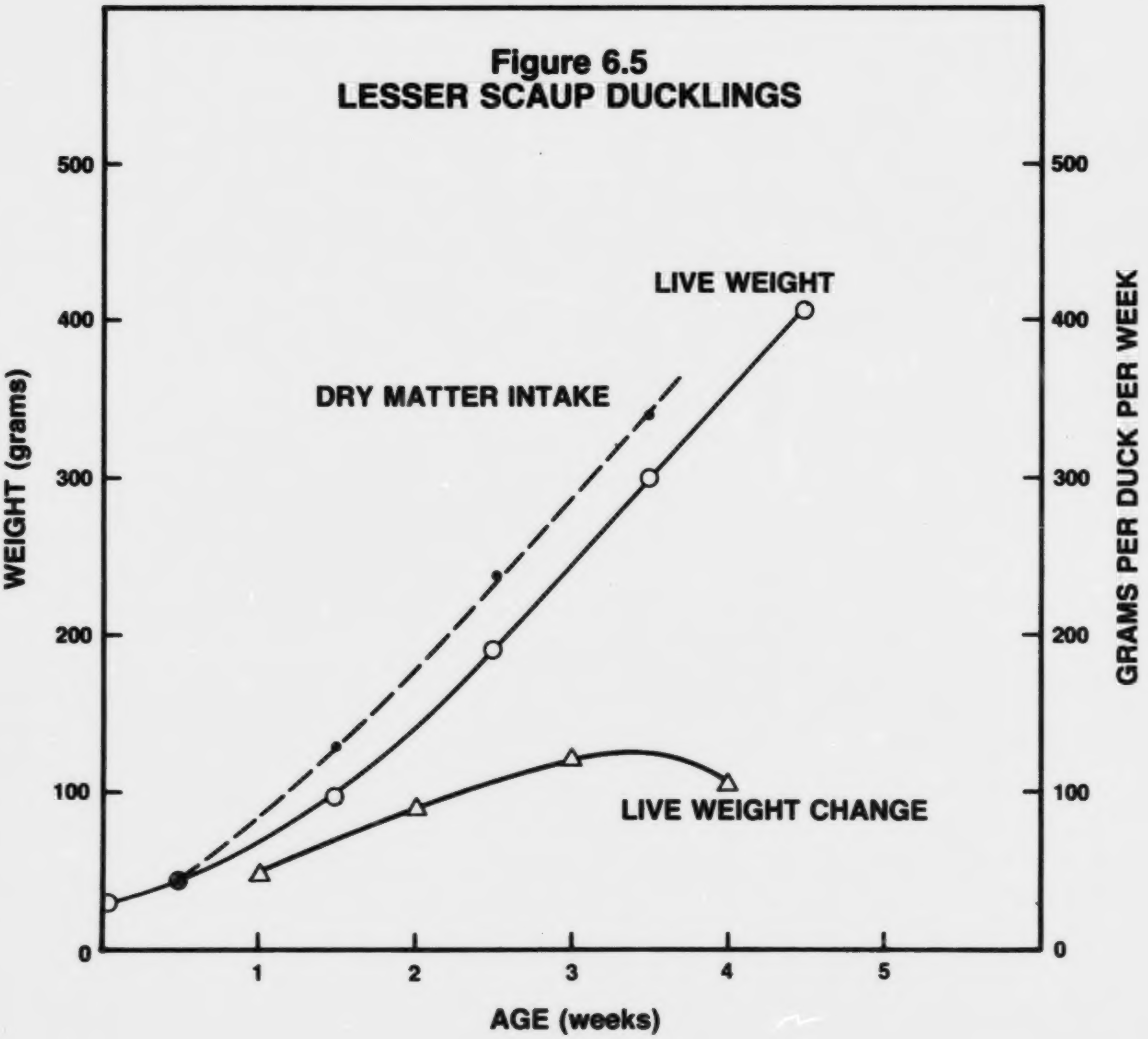


Figure 6.5
LESSER SCAUP DUCKLINGS

Mallard growth

Weight is related to age for weeks one to four through the following regression equation:

$$\text{Weight (g)} = 187.55 \text{ Age (weeks)} - 86.18, \ r = .99 \tag{4}$$

Dry matter intake is related to age for weeks one to four through the following regression equation:

$$\text{Dry Matter Intake (g dry wt. week}^{-1}) = 109.40 \text{ Age (weeks)} + 60.43, \ r = .98 \tag{5}$$

Growth is calculated from the ratio of live weight change (LWC) to dry matter intake (DMI) as a function of age:

$$\frac{\text{LWC (g live wt. week}^{-1})}{\text{DMI (g dry wt. week}^{-1})} = -\,0.10 \text{ Age (weeks)} + 0.80, \ r = -.99 \tag{6}$$

Energetic costs in terms of dry matter intake at a specific age is estimated as:

$$\text{Energetic costs} = \left| 1 - \frac{\text{LWC}}{\text{DMI}} \right| \times \text{DMI} \tag{7}$$

Mallard ducklings are assumed to feed exclusively on aquatic invertebrates until at least two weeks of age.

Lesser Scaup growth

Weight is related to age for weeks one to four through the following regression equation:

$$\text{Weight (g)} = 101.50 \text{ Age (weeks)} - 58.08, \ r = .99 \tag{8}$$

Dry matter intake is related to age for weeks one to four through the following regression equation:

$$\text{Dry Matter Intake (g dry wt. week}^{-1}) = 104.00 \text{ Age (weeks)} - 23.67, \ r = .99 \tag{9}$$

Growth is calculated from the ratio of live weight change to dry matter intake as a function of age:

$$\frac{\text{LWC (g live wt. week}^{-1})}{\text{DMI (g dry wt. week}^{-1})} = -0.095 \text{ Age (weeks)} + .675, \ r = .99 \tag{10}$$

Energetic costs in terms of dry matter intake are estimated as with the Mallard. Lesser Scaup ducklings are assumed to feed exclusively on aquatic invertebrates until at least four weeks of age.

The relationship between consumption and prey density was developed from work by Sugden (1971) who measured feeding rates for three week old Lesser Scaup ducklings in laboratory tanks under conditions of of constant prey (Gammarus) density. He reported that the consumption rate increased linearly with increasing Gammarus densities from 25 to 200 ind. m^{-3} (0.1 to 0.8 g dry wt. m^{-3}). The number of prey taken per dive increased linearly with increased prey density, however, the number of dives increased with density only between 25 and 50 ind. m^{-3} and then plateaued at >4 ind. min^{-1}. Below prey densities of 25 per m^3 the behavioural reinforcement for feeding was weak.

The consumption rate for feeding ducklings can be expressed in terms of prey density as follows:

$$\text{Consumption Rate (prey min}^{-1}) = .165 \text{ Prey Density (prey m}^{-3}) + 3.465 \qquad r = .95 \qquad (11)$$

or

$$\text{Consumption Rate (mg dry wt. min}^{-1}) = .165 \text{ standing biomass (mg dry wt. m}^{-3}) + 13.86 \qquad (12)$$

Based on estimated energy requirements, Sugden (1971) concluded that 25 ind. m^{-3} averaging 4 mg dry wt. per ind. (~12 calories per individual) was insufficient to support a three-week-old duckling even if it fed continuously for 24 hours. He estimated that a duckling could conceivably obtain sufficient food feeding approximately 40% of the time at a prey density of 50 ind. m^{-3} (0.2 g dry wt. m^{-3}). At 50 ind. m^{-3}, approximately 12 prey are consumed per minute according to the regression (11) from Sugden's data. Pehrsson (1979) estimated that wild Mallard ducklings less than 20 days old captured between 11 and 13 prey items per minute in productive lakes without fish.

The estimate of 4 mg average dry weight of prey used by Sugden (1971) may be appropriate for aquatic invertebrate prey in prairie potholes according to mean dry weight data for taxa (Driver _et al_. 1974) and taxonomic distributions of organisms in seasonal and semipermanent ponds (see Section 2.3). Although there are no reports on body size and weight distributions of invertebrates in prairie potholes, Armstrong and Nudds (1985) reported a high frequency of large invertebrates 7 to 12 mm in length associated with aquatic vegetation in duck feeding areas.

Ducklings between one and four weeks of age under natural and captive conditions spend between 40-60% of their time on the water feeding (E. Driver CWS Prairie Migratory Bird Research Centre, pers. comm.). As visual predators, most feeding is assumed to be during daylight hours, although Swanson and Sargeant (1972) reported that mature birds and ducklings may feed actively at night during periods of insect emergence. For the model, time spent feeding is assumed to be constant and equal to 10 h daily, a value representing 60% of the nearly 16 h of summer sunlight. There is some data to suggest that ducklings may feed longer on food-depleted aquatic ecosystems (J.-L. Desgranges CWS Quebec Region, pers. comm.). Hunter _et al_. (1984) reported that broods on carbaryl-treated ponds spent approximately 10% more time searching for food than did a brood on an uncontaminated pond. More time spent searching for prey means greater energetic costs expended by ducklings. However, as the model assumes regardless of food supply, the assumption of a constant daily feeding period may help balance the assumption of constant maintenance costs.

Assumptions in the DUCKLING GROWTH MODELS are as follows:

1. There is a direct relationship between aquatic macroinvertebrate density/standing biomass and duckling growth and survival.

2. Mallard ducklings consume only aquatic invertebrates for the initial two weeks of growth and Lesser Scaup ducklings are entirely dependent on aquatic invertebrates during the first four weeks of growth.

3. Growth rate in captive ducklings fed *ad libitum* under laboratory conditions is approximately equal to the optimal growth rate of ducklings in the wild.

4. Consumption rate is related to prey density/biomass according to the linear regressions (11) or (12).

5. Time spent feeding by ducklings is constant and equal to 10 h daily.

6. Potential dietary intake is directly related to age as in equations (5) and (9).

7. Energetic costs (g week^{-1}) are constant regardless of available food supply and decrease with the age and weight of the duckling between the second and fourth week.

If consumption rate is constant over the feeding period and standing invertebrate biomass is not readily depleted by feeding activity:

$$\text{Dry Matter Intake (g dry wt. per week)} = 4200 \text{ (min. week}^{-1}\text{ feeding time)} \times \text{consumption rate (mg dry wt. min}^{-1}\text{)} \qquad (13)$$

In order to predict the macroinvertebrate standing crop necessary to support a specific growth rate, consumption rate (dry matter intake) is used to estimate macroinvertebrate standing crop based on the previously stated assumptions:

$$\text{Macroinvertebrate Standing Biomass (g dry wt. m}^{-3}\text{)} = 0.0013 \text{ DWI (g dry wt. m}^{-3}\text{)} - .044 \qquad (14)$$

and

$$\text{Macroinvertebrate Density (members m}^{-3}\text{)} = 0.329 \text{ DWI (g dry wt. m}^{-3}\text{)} - 10.99 \qquad (15)$$

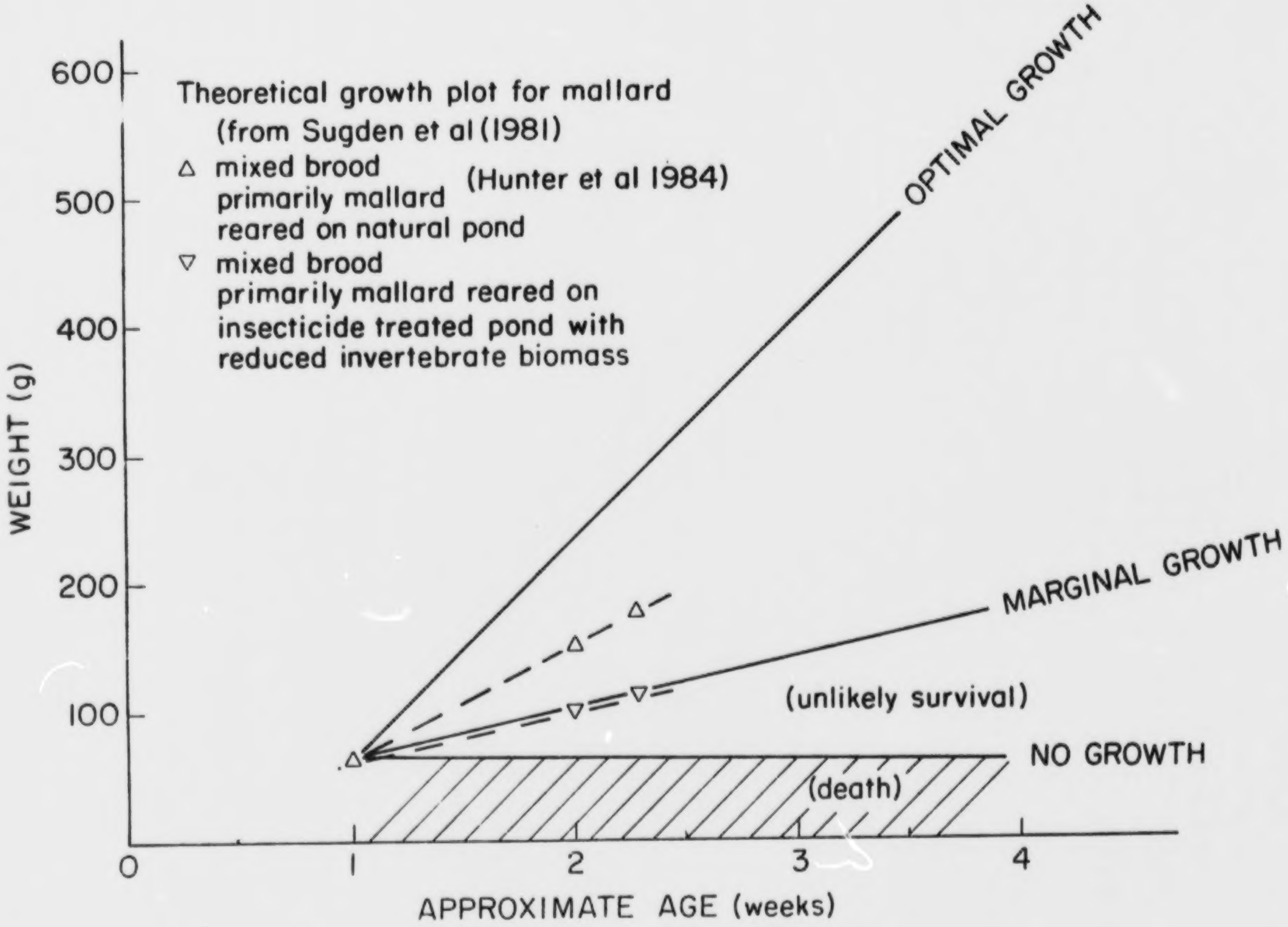
Two examples of predictions from the DUCKLING GROWTH MODELS are presented in Table 6.23 and plotted for Mallards in Figure 6.6. The optimal growth rate for a two-week-old Mallard, 162 g week^{-1}, can be supported by a continuous supply of macroinvertebrates at approximately 80 ind. m^{-3} (0.32 g dry wt. m^{-3}). Approximately 39 ind. m^{-3} (0.16 g dry wt. m^{-3}) are required to support marginal growth (25% of potential) of 42 g week^{-1}, which is similar to the growth estimated by Hunter *et al*. (1984) for broods on a

Table 6.23. **Predicted macroinvertebrate standing crop necessary to support various levels of growth rate for Mallard and Lesser Scaup ducklings.**

	Dry Matter Intake	Growth Rate	Macroinvertebrate Standing Crop Necessary to Support Growth Rate	
	(g dry wt.week^{-1})	(g live wt.week^{-1})	(numbers m^{-3})	(g dry wt. m^{-3})
Mallard, 2 weeks old (<285 g)	279	167 (optimal)	80	0.32
	152	42 (marginal)[a]	39	0.16
	112	0 (no growth)	26	0.10
Lesser Scaup, 4 weeks old (<348 g)	392	116 (optimal)	118	0.47
	305	29 (marginal)[a]	89	0.36
	276	0 (no growth)	80	0.32

[a] Marginal growth is defined as approximately 25% of potential growth.

Figure 6.6. Plot of growth for Mallard ducklings based on the DUCKLING GROWTH MODEL and data from Sugden _et al_. (1981). Live weight of predicted growth at one week was shifted to align with duckling weight from the Hunter _et al_. (1984) study.



600
500
400
300
200
100
WEIGHT (g)
Theoretical growth plot for mallard
(from Sugden et al (1981)
△ mixed brood primarily mallard reared on natural pond
(Hunter et al 1984)
▽ mixed brood primarily mallard reared on insecticide treated pond with reduced invertebrate biomass
OPTIMAL GROWTH
MARGINAL GROWTH
(unlikely survival)
NO GROWTH
(death)
0
1
2
3
4
APPROXIMATE AGE (weeks)

carbaryl stressed pond (Figure 6.6). At macroinvertebrate densities below 26 ind. m^{-3} (0.10 g dry wt. m^{-3}), no growth or weight loss is predicted to occur.

The optimal growth rate for a four-week-old Lesser Scaup is less than that of the two-week-old Mallard due to additional maintenance costs. The estimated standing crop of macroinvertebrate prey necessary to support the Scaup (112 ind. m^{-3}, 0.47 g dry wt. m^{-3}) is approximately 1.5 times the food resource base of the two-week-old Mallard. Marginal growth requires 89 ind. m^{-3} or .36 g dry wt. m^{-3} and maintenance requires 80 ind. m^{-3} or 0.32 g dry wt. m^{-3}.

The model suggests that a continuous supply of prey at a standing biomass of approximately 4.2 g dry wt. m^{-3} would be necessary to support optimal growth for an average brood of nine Lesser Scaup ducklings four weeks of age. Based on data summarized in Section 2.3, 4.2 g dry wt. m^{-3} is near the middle of the range of reported average values. Gerking (1962) estimated that the production of benthic fauna in a shallow lake during summer months is twice the standing biomass. Based on a production turnover ratio of two per month or approximately 0.5 per week, approximately 2.1 g m^{-3} invertebrate biomass can be generated within a week. If we assume that duckling feeding equals production and therefore does not deplete the resource supply, the brood would have to search approximately 1715 m^3 of pond per week to obtain the necessary dry matter intake for optimal growth. This estimate of habitat food capacity appears reasonable considering the dimensions of semi-permanent potholes preferred by Scaups. Approximately 1200 m^3 of pond habitat would be required for optimal growth of a Mallard brood of nine ducklings two weeks of age under the same conditions as described above for the Lesser Scaup.

A crude estimate of the available standing crop biomass in the carbaryl treated ponds is approximately 0.4 g dry wt. m^{-3} based on extrapolation of data from Gibbs _et al_. (1984). Based on the estimated 0.32 g dry wt. m^{-3} necessary to support a single duckling at the rate of optimal growth, available resources after the invertebrate kill were far short of the requirements for the seven-member test brood. The brood on the treated pond would be required to search nearly 5400 m^3 $week^{-1}$ to obtain sufficient food to grow at 25% of their potential.

Relating predictions of necessary macroinvertebrate biomass from the DUCKLING GROWTH MODELS to insecticide stress may be crudely done with the benchmarks in Table 6.22 assuming an average macroinvertebrate standing biomass of approximately 4.0 g dry wt. m^{-3}. Compounds with a RHI equivalent to that of carbaryl should reduce standing crop levels of macroinvertebrates in productive potholes to less than 1.0 g dry wt. m^{-3}. This prey resource level is predicted to be insufficient to support marginal growth in an average brood of six Mallards or Lesser Scaups in a pothole of less than 5500 m^3. Compounds calculated to have a relative hazard equivalent to permethrin are expected to deplete macroinvertebrate standing stocks by >90%, reducing expected biomass to <0.4 g dry wt. m^{-3}. This level is predicted to be just sufficient to support marginal growth in a single Lesser Scaup or Mallard duckling and totally inadequate for a normal size brood of one week age in a pond with anything less than 2500 m^3 of productive littoral habitat.

6.3.13 Critique of secondary impact model applied to ducklings

The model at present is in an early stage of development. The accuracy of predictions is difficult to evaluate as no study is available which experimentally defines carrying capacity of an aquatic ecosystem for ducklings in terms of any sampling measure of macroinvertebrate abundance. Perhaps current experimental programs at the Northern Prairie Wildlife Research Centre will partially fill this information gap. Waterfowl biologists agree that ducklings are themselves the best samplers of food resources, but they do not agree on which quantitative sampling techniques best estimate available resources. In this model we have focused on macroinvertebrate populations associated with littoral macrophytes, as being most representative of available prey for ducklings. However, ducklings have different feeding habits correlated with age, species and the ecological characteristics of the aquatic system. They are also somewhat generalist in habit and take what is available and can be caught and eaten. Therefore, the most obvious and critical problem area in the current model development is quantifying prey availability and prey size selection in aquatic systems.

The bioenergetics approach does offer several advantages in estimating food resource requirements to support duckling growth:

- calculation is simple;

- data requirements are minimal;

- predictions of prey biomass to support optimal, marginal or no growth rates appear reasonable;

- a similar model may be developed for clutch size as a function of food resources for the laying hen.

There are also a number of problem areas other than the critical issue of correctly quantifying prey availability:

- consumption-availability relationships are based solely on data for a single species and prey item: Lesser Scaup feeding on Gammarus;

- the assumption of a constant 10 h daily feeding period may be unrealistic under conditions of reduced available food;

- in turn, the assumption of constant energetic costs of maintenance under both conditions of food sufficiency and insufficiency may be unrealistic;

- model predictions are highly dependent on benchmarks of the RHI which, at present, provide only crude resolutions of impact.

Further model refinements will be highly dependent on new data providing benchmarks with greater resolution power and further empirical information on the relationship between macroinvertebrates, standing crops and duckling growth (and survival).

6.4 Threat of Synthetic Pyrethroids and Other Insecticides to Waterfowl

In summation, laboratory toxicity data, field test data and the associated hazard analysis suggest that a number of highly toxic insecticides may present a substantial risk to waterfowl. Although there is the risk of direct effects with certain compounds, specifically carbofuran and diazinon, indirect effects mediated through reductions in available food resources are more probable. This conclusion is particularly important since it emphasizes that hazard evaluators should consider both potential for direct and indirect (ecologically mediated) effects from insecticides. The case for ducklings may be stated as follows: 1) ducklings feed almost exclusively on aquatic invertebrates in their first two weeks of life (or longer for some species); 2) this critical growth period can coincide with pesticide spraying in overlapping areas of agriculture and breeding habitat; 3) impacted brood rearing ponds would experience widespread severe reductions in invertebrate fauna following overspray of relatively high hazard insecticides such as the pyrethroids, azinphos methyl, chlorpyrifos and malathion; 4) the impacts on invertebrate abundance are expected to persist for most of the summer and well beyond the critical period of growth in prefledgling ducklings; and 5) data on adult and duckling feeding indicates that insufficient invertebrate food can cause reduced clutch size in laying females as well as mortality and severe reductions in growth of young ducklings. The response is rapid. The evidence above suggests the following testable hypothesis:

HO: No duckling mortality and/or growth reduction during the phase of critical dependence on aquatic invertebrates will occur due to pesticide reduction of essential invertebrate food items.

HA: Pesticide destruction of the aquatic invertebrate community during periods of duckling dependency will result in significant growth reduction and possible mortality in prefledglings.

Two partial tests of the above hypothesis suggest that indirect impacts reducing duckling well-being are indeed probable (Hunter et al. 1984, G. Shaw CWS Western and Northern Region, pers. comm.).

Properly designed and executed field studies are essential to quantify the pesticide-invertebrate-duckling relationship under relevant environmental conditions and to calibrate model predictions. An intensive long-term monitoring study of duckling recruitment during "normal" and "high" spray years is needed to quantify the extent and severity of the potential threat of aerial insecticide spraying in the prairie pothole region. The ecological risk of the synthetic pyrethroid insecticides appears particularly high due to their extreme and largely non-selective toxicity to aquatic invertebrates and potential long-term persistence in the sediments. Unrestricted aerial application of these compounds in areas of waterfowl breeding habitat is unacceptable without field studies demonstrating low or no impact on clutch size or duckling success. Re-evaluation of the aerial use, in the potholes region, of other insecticides rated as highly hazardous (chlorpyrifos, azinphos methyl) may be advisable. Refinements in field test guidelines are required prior to further evaluation of insecticide compounds with high aquatic invertebrate toxicity and a use pattern of aerial application on prairie crops.

Efforts are needed to put the potential stress from insecticides in perspective with other forms of stress on breeding waterfowl populations such as physical habitat destruction. This objective will require both modelling and broadscale field monitoring of recruitment success.

Appendix 6.1. Physico-chemical parameters used for persistence and risk calculations.

Table 1. Input parameters for the synthetic pyrethroids.

	Cypermethrin	Deltamethrin	Fenvalerate	Permethrin
Melting point (°C)	0[a]	101	0[a]	0[a]
Molecular weight	416	503.2	420	391
Vapor pressure (torr)	1.2×10^{-8b}	1.5×10^{-8}	2.8×10^{-7} (at 25°C)	3.4×10^{-7} (at 25°C)
Octanol-water partition coefficient	29500	270000	26,500	170000
Water solubility (ppm)	0.041	0.01	0.085	0.04
Quantum yield	0.12	0.28	0.08	0.51
Fraction biodegradable	0.1	0.1	0.1	0.1
First order hydrolysis rate constant	3.4×10^{-3} at pH 6 9.9×10^{-3} at pH 8	3.4×10^{-3} at pH 6[c] 9.9×10^{-3} at pH 9[c]	7.28×10^{-4} at pH 6 7.38×10^{-3} at pH 8	3×10^{-7} at pH 5 3×10^{-3} at pH 9
Molar extinction coefficient at:				
Y 290	433.1	680	596	76.7
300	-	-	-	-
310	-	-	-	-
320	-	-	-	-
330	-	-	-	-
340	-	-	-	-
350	-	-	-	-
360	-	-	-	-
370	-	-	-	-
380	-	-	-	-
390	-	-	-	-
400	-	-	-	-

[a] Melting point is entered as 0°C if the compound is a liquid at system temperature

[b] Estimated assuming that the heat of vaporization of cypermetherin is equivalent to that of permethrin

[c] Estimated to be the same as cypermethrin

Appendix 6.1. (cont'd)

Table 2. Input parameters for insecticides other than the synthetic pyrethroids.

	Azinphos Methyl	Carbaryl	Carbofuran	Chlorpyrifos	Diazinon
Melting point (°C)	73	142	152	43.5	84
Molecular weight	317.3	201	221.3	350.5	304.36
Vapor pressure (torr)	3.8×10^{-4} (at 20°C)	5×10^{-3} (at 26°C)	6.84×10^{-7} (at 20°C)	4.62×10^{-6} (at 20°C)	1.4×10^{-3} (at 20°C)
Octanol-water partition coefficient	0[a]	197	40	66000	1386
Water solubility (ppm)	22	104	415	0.4	40
Quantum yield	0[a]	0.006	0[a]	0[a]	0[a]
Fraction biodegradable	0.1	0.1	0.1	0.1	0.1
First order hydrolysis rate constant	0 at pH 5 6.1×10^{-2} at pH 9	4.6×10^{-4} at pH 5 4.6 at pH 9	3.4×10^{-4} at pH 5 1.19 at pH 9	2.1×10^{-2} at pH 5 4.33×10^{-2} at pH 9	2.2×10^{-2} at pH 5 5.0×10^{-3} at pH 9
Molar extinction coefficient at:					
γ 290	1[a]	3100	1[a]	1[a]	1[a]
300	1	1100	1	1	1
310	1	320	1	1	1
320	1	110	1	1	1
330	1	11	1	1	1
340	1	0	1	1	1
350	1	0	1	1	1
360	1	0	1	1	1
370	1	0	1	1	1
380	1	0	1	1	1
390	1	0	1	1	1
400	1	0	1	1	1

Appendix 6.1. (cont'd)

Table 2 (cont'd)

	Dimethoate	Malathion	Methoxychlor	Phosmet
Melting point (°C)	52	0^b	89	72.9
Molecular weight	229.3	330.36	345.5	317.3
Vapor pressure (torr)	7.95×10^{-6} (at 20°C)	4×10^{-5} (at 20°C)	1.24×10^{-6} (at 25°C)	1×10^{-3} (at 50°C)
Octanol-water partition coefficient	513	781	47,500	677
Water solubility (ppm)	25000	145	0.04	25
Quantum yield	0^a	0^a	0.3	0^a
Fraction biodegradable	0.1	0.1	0.1	0.1
First order hydrolysis rate constant	5×10^{-3} at pH 5 2.5×10^{-1} at pH 9	6.6×10^{-4} at pH 5 6.6 at pH 9	1.9×10^{-3} at pH 7 2.2×10^{-3} at pH 9	9×10^{-2} at pH 5 6.3 at pH 9
Molar extinction coefficient at:				
γ 290	1^a	1^a	100	1^a
300	1	1	2	1
310	1	1	.1	1
320	1	1	-	1
330	1	1	-	1
340	1	1	-	1
350	1	1	-	1
360	1	1	-	1
370	1	1	-	1
380	1	1	-	1
390	1	1	-	1
400	1	1	-	1

[a] Default value

[b] Melting point is entered as 0°C if the compound is a liquid at system temperature

CHAPTER 7

TOXICITY OF HERBICIDES IN SLOUGH ECOSYSTEMS AND UPLAND HABITAT: INDIRECT THREAT TO WATERFOWL

Herbicides have been shown to affect the success of wildlife through critical alterations in habitat. A recent review of the impacts on wildlife due to herbicide-induced changes in the vegetative structure of forests pointed out that a primary factor governing the distribution and abundance of wildlife is the structure and composition of vegetation (Morrison and Meslow 1982). The richness and diversity of the animal community can often be related directly to the vertical and horizontal diversity of vegetation. As herbicides induce vegetation changes, subsequent changes in the associated animal community are expected.

A well documented example of the indirect impact of herbicides on an avian species is the recorded reduction (probably 10-15%) in Grey Partridge (Perdix perdix) populations in English farmlands (Potts 1974, 1977, 1985). A reduction in chick survival was the principal cause of the Grey Partridge decline and was caused, in turn, by the depletion of the quality and quantity of plant and invertebrate food resources for chicks resulting from herbicide use (Southwood & Cross 1969). From 1952 through 1976, with the increasing and widespread use of herbicides (and insecticides) on cereal crops, the survival rate for Partridge chicks was halved: chick mean survival rate was 50% for all years censused (1903-39, 1947-50) prior to chemical applications for weed control and dropped to <25% (Potts 1985). Starvation was the main cause of chick loss. Potts experimentally demonstrated that the use of herbicides in the removal of broadleaf weeds resulted in the reduction of preferred insect food resources by about 50%, a magnitude sufficient by itself (without accounting for the effects of insecticides) to account for the reported reductions in chick survival (Potts 1985). Equally as important, large-scale controlled and replicated experiments reducing herbicide use produced an immediate and significant increase in Partridge chick survival (Rands 1985). Although the downward trend in chick survival has been substantial, annual variation was high so differences were significant only if monitored over many years. The long period of intensive monitoring required is one of the main problems in demonstrating the indirect impacts of pesticides on wildlife.

Although there is insufficient data to directly correlate pesticide use with the declining success of waterfowl on their breeding grounds, there is evidence to suggest that the extensive herbicide application in the pothole region of the Canadian prairies may adversely alter food, nesting and protective cover for some resident duck species. Field studies involving herbicide impacts and waterfowl response are generally unavailable and analysis is, therefore, strongly based on extrapolation of ecological relationships between habitat resources and waterfowl species (see Section 2.5) to situations that may exist under conditions of herbicide stress. Herbicide-induced changes are expected to provoke a range of responses because waterfowl species run the continuum from specialist to generalist in use of habitat resources.

In this chapter we examine the factors which control the distribution and persistence of herbicides in sloughs and their margins. The ramifications of herbicide sequestering is analyzed with respect to exposure patterns for slough flora and fauna. Where there are indications of vegetative changes resulting from major use herbicides that may degrade the quality or diminish the quantity of 1) plant or animal food resources, 2) nesting material or cover, or 3) protective cover, we attempt to assess the response of various waterfowl species to these resource disruptions.

Ecological changes within prairie sloughs due to the indirect contamination of herbicides are not expected to be dramatic. However, subtle changes at the gross vegetative level can produce more substantial changes in floral composition and faunal make-up. Much of the data produced in controlled field studies is based on high exposure levels more consistent with aquatic weed control than with the combination of overspray and runoff contamination expected in prairie sloughs (Chapter 5). As exceedingly few herbicide studies have incorporated waterfowl, their response to herbicide-induced changes in slough habitat must be largely extrapolated from what is known of species habitat preferences. The ecotoxicological assessment of herbicide hazards is thus more strongly dependent on projections of ecological interaction at expected contaminant concentrations than was the case for the indirect insecticide impacts developed in the previous chapter.

7.1 Expected Exposure Pattern in Herbicide-Contaminated Sloughs

The pattern of direct toxic response of aquatic invertebrates to insecticides, followed by subsequent adverse impacts on the feeding of ducks at critical life stages, can generally be explained in terms of an acute exposure scenario, although chronic exposure patterns may affect the recovery of the biological community. Herbicide use patterns and environmental stability suggest that a greater emphasis be placed on chronic exposure patterns when assessing herbicide hazards to waterfowl within the prairie pothole system. Unlike insecticides which are applied sporadically for pest control, herbicides for weed control are applied regularly (applications often repeated annually) and this type of use pattern leads more readily to conditions of residue build-up. Therefore, herbicide persistence in pond water, macrophytes and sediment as well as in slough margins is particularly critical to determining long-term exposures and subsequent chronic effects. Although the connection between phytotoxic agents and the resources assuring duck population success is not as evident as with insecticides, the long-term continuous annual spraying, which reduces margin and upland cover is probably of great importance to duck nesting and brood success. Because of the complexity of the network of probable events from herbicide contamination of agricultural sloughs to the diminished success of various duck species, exposure assessment is particularly critical in this hazard analysis. Exposure patterns must be illuminated in order to determine appropriate probability estimates of an adverse impact.

Although the need for data on the partitioning and persistence of herbicides in sloughs or pond-like ecosystems is obviously critical to a through hazard evaluation, data are available only for those few compounds which are used for aquatic macrophyte control (review by Muir 1986). More is generally known about herbicide fate in soils and vegetation, although not with specific reference to pothole margins or upland areas.

Figure 7.1 illustrates that persistent compounds applied annually can build up in soils (and sediments). Theoretically, if 80% of applied quantities were lost per year through degradation and transfer processes, a maximum residue of 25% of the application rate would build up in three years. However, if only 50% of a compound is lost between annual applications, residue levels approach nearly 90% of application rates within three years. Table 7.1 summarizes data compiled by Smith (1982) which indicate that triallate, trifluralin and linuron may be carried over to subsequent years at levels 25% or more of the applied rate. Fall application in the cold Canadian climate tends to enhance carry-over. Even applications of 2,4-D, which is normally rapidly degraded, may remain in agricultural soils into the next growing season if applied in the fall (Nicholaichuk & Grover 1983). An extreme example of persistence leading to residue build-up is the herbicide tebuthiuron. When applied at 840 g AI ha^{-1}, 38% of applied tebuthiuron remained in field soils after 21 months and was still effective in controlling target weed species (Emmerich et al. 1984). The period required to reach non-detectable levels of tebuthiuron was estimated to be from three to seven years. Atrazine persisted into the second year in water and sediments of a treated (300 $\mu g\ L^{-1}$) farm pond at levels of approximately 10% of the application rate (Klaassen and Kadoum 1979). This example is particularly pertinent as carry-over was demonstrated under conditions of normal application rates in a shallow pond. All examples strongly support the need to examine herbicide stability and persistence in slough ecosystems and to consider the ramifications of the accumulation of residues of stable compounds which are often applied year after year for weed control.

7.1.1 Factors affecting herbicide fate in pothole ecosystems

The fate of herbicides, once introduced into aquatic ecosystems, is governed by various physical (adsorption, volatilization), chemical (hydrolysis, photolysis) and biological processes (biodegradation). A concise description of these processes as related to herbicide fate is presented in Muir (1986). Only a brief discussion relating these processes to shallow ponds is attempted here.

Herbicide sorption to aquatic sediments and seston parallels soil sorption. Adsorption of most neutral nonionizable herbicides has generally been described by the Freundlich relationship where the partition coefficient (Kp) is often assumed to equal the ratio of the concentration in sediment ($\mu g\ g^{-1}$) to the concentration in water ($\mu g\ mL^{-1}$). Kp values for certain compounds (e.g. triazines) correlate well with the organic content of sediment, expressed as K_{OC}. High values of K_{OC} indicate

Figure 7.1. Hypothetical plot of build-up of herbicide residues in soil and sediment based on percent yearly carry-over of the undegraded portion of the compound.

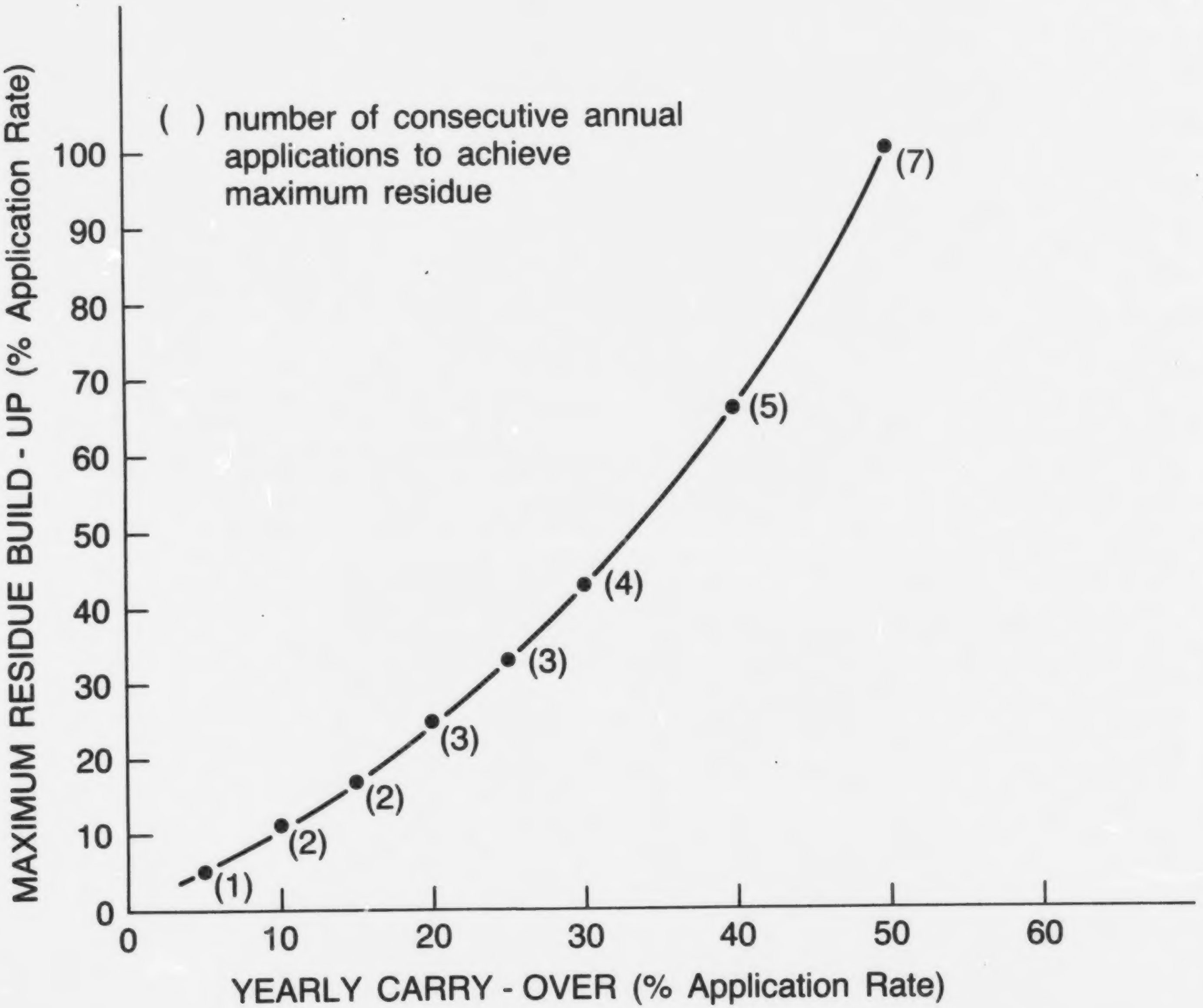

( ) number of consecutive annual applications to achieve maximum residue
MAXIMUM RESIDUE BUILD - UP (% Application Rate)
0
10
20
30
40
50
60
70
80
90
100
YEARLY CARRY - OVER (% Application Rate)
0
10
20
30
40
50
60
(1)
(2)
(2)
(3)
(3)
(4)
(5)
(7)

Table 7.1. Persistence of select herbicides in Canadian agricultural soils[a].

Treatment	Application Rate (kg/ha)	Carry-over[b] (% of applied)	Reference
		Acid Herbicides	
		Dicamba	
August, field plots	1.1-4.5	Less than 5% after 7 wks	Stewart and Gaul 1977
May, small-plot study	2.2	None detected after 22 wks	Smith and Hayden 1976
July, field plots	0.56-1.12	26-32% after 48 wks, about 14% after 104 wks	Keys and Friesen 1968
September, field plots	1.62-3.24	Cereal damage after 32 and 84 wks	Vanden Born 1969
June, 15 cm incorporation in field plots	4.48	66% after 20 wks, 35% after 48 wks and 6% after 99 wks	Ragab 1975
June, field plots	0.017-0.07	6-18% after 17 wks	Hunter and Stobbe 1972
June, field plots	0.035-0.7	70-92% after 17 wks, 32-69% after 52 wks	Hunter and Stobbe 1972
October, field plots	0.56-4.48	20-30% after 52 wks, 10% after 104 wks	Keys and Friesen 1968
		Carbamate Herbicides	
		EPTC	
May, 5 cm incorporation in small plots	4.0	0-8% after 22 wks	Smith and Hayden 1982
		Triallate	
May, 5 cm incorporation in small plots	2.8	16-27% after 22 wks	Smith 1971
May, 5 cm incorporation in small plots	1.7	3-35% after 22 wks, 5-25% after 52 wks, 0-12% after 72 wks	Smith and Hayden 1976

Table 7.1 (cont'd)

Treatment	Application Rate (kg/ha)	Carry-over[b] (% of applied)	Reference
May, 5 cm incorporation in small plots	1.5	12-23% after 20 wks	Smith 1979
October, 5 cm incorporation in small plots	1.7	23-75% after 30 wks	Smith 1975
		Dinitroanaline Herbicides	
		Trifluralin	
May, 10 cm incorporation in field plots	Several	17-26% after 50 wks	Pritchard and Stobbe 1980
May, 10 cm incorporation in field plots	0.75	44-74% after 28 wks	Jensen and Kimball 1980
November, 10 cm incorporation in field plots	1.12-2.24	Phytotoxic residues after 12 wks, none after 70 wks	Rahman and Ashford 1973
May, 5 cm incorporation in small plots	1.4	11-21% after 22 wks	Smith 1972
May, 5 cm incorporation in small plots	1.1	8-31% after 22 wks, 5-35% after 52 wks, 2-15% after 72 wks	Smith and Hayden 1976
May, 5 cm incorporation in small plots		9-20% after 20 wks	Smith 1979
May, 5 cm incorporation in small plots	1.0	7-29% after 20 wks, 6-20% after 52 wks	Hayden and Smith 1980
October, 5 cm incorporation in small plots	1.1	31-71% after 30 wks	Smith 1975
		Phenoxy Herbicides	
		2,4-D	
June, a field study with a butyl ester	0.42	None detected after 4 wks	Cochrane and Russell 1975

Table 7.1 (cont'd)

Treatment	Application Rate (kg/ha)	Carry-over[b] (% of applied)	Reference
August, a field study with amine and ester formulations	5.6-7.8	Less than 5% after 10 wks, none detected after 52 wks	Stewart and Gaul 1977
May, small plot study with acid	2.2	None detected after 22 wks	Smith and Hayden 1976
		Triazine Herbicides	
		Atrazine	
June, field plots	2.24-22.4	18-44% after 10 wks, 6-8% after 52 wks	Birk and Roadhouse 1964
May-June, field treatments	1.12-3.36	About 15% after 22 wks, less than 10% after 52 wks	Sirons et al. 1973
May-June, field plots	0.28-2.24	10-20% after 52 wks	Marriage et al. 1978
May-June, field treatments	1.68	48-53% after 7 wks, 23-28% after 18 wks, about 10% after 52 wks	Khan et al. 1981
		Cyanazine	
May-June, field treatments	1.12-3.36	Less than 5% after 22 wks	Sirons et al. 1973
June, field plots	2.0	About 15% after 14 wks	Yoo et al. 1981
		Metribuzin	
June-August, field plots	1.0-2.0	Less than 10% at end of growing season	Webster and Reimer 1976
May-June, field plots	1.12	About 10% after 12 wks	Marriage et al. 1978
May, small plot study	1.0	0-20% after 22 wks	Smith and Hayden 1982
		Simazine	
June, field plots	2.24-4.48	About 14% after 52 wks	Ragab and Leefe 1972

Table 7.1 (cont'd)

Treatment	Application Rate (kg/ha)	Carry-over[b] (% of applied)	Reference
June, 15 cm incorporation in field plots	4.48	40% after 9 wks, 18% after 23 wks	Ragab 1974
May, field plots	0.56-22.4	About 35% after 14 wks, about 20% after 52 wks	Roadhouse and Birk 1961
May, small-plot study	4.5	14-60% after 22 wks, 10-53% after 52 wks, 9-30% after 72 wks	Smith and Hayden 1976
		Urea Herbicides	
		Linuron	
Field applications	2.24	Crop damage after 52 wks	Morris and Penney 1971
May, small-plot study	2.2	28-42% after 28 wks, 27-32% after 52 wks, 3-18% after 72 wks	Smith and Emmond 1975
May, field plots	2.24-4.48	59-74% after 17 wks, 14-21% after 60 wks	Khan *et al.* 1976

---

[a] Modified from Smith 1982

[b] Normally measured to a depth of 10 to 15 cm

partitioning to sediments and low K_{OW} values indicate retention in water. Empirical relationships between K_{OC} and K_{OW} for a number of hydrophobic herbicides have been reported (Rao & Davidson 1982). Organic content of pothole sediments is expected to be relatively high, between 10-20% (Section 2.2.2), which means that a substantial portion on nonionizable herbicides will be sequestered on sediment, seston and macrophyte surfaces.

Adsorption of cationic herbicides to sediment is best described by Langmuir isotherms where the partition coefficient (K_{LM}) and adsorption capacity (X_c)can be determined from a plot of the ratio of concentration in water to concentration in sediment against concentration in water (see Bowmer 1982a). The sorption of diquat (Corwin & Farmer 1984), paraquat (Karickoff & Brown 1978) and glyphosate (Bowmer 1982a) to sediment and suspended material have been adequately described by Langmuir isotherms. The clay mineral content of pothole sediment is expected to be a factor in herbicide adsorption in areas where agricultural soils have a high clay content.

Desorption is a critical process affecting the extent of the release of herbicides from sediment. Although desorption is frequently assumed to be the reverse of adsorption, both neutral and cationic herbicides exhibit hysteresis (Corwin & Farmer 1984).

Volatilization can be a major pathway of loss of some herbicides from shallow potholes. The transfer process is dependent on the Henry's constant of the chemical and the turbulence in the water and atmosphere. Henry's constant is influenced by both solubility and vapour pressure. The greatest volatile losses occur with compounds which are hydrophobic and have high vapour pressures. However, the low vapour pressure of some hydrophobic compounds may be offset by high activity coefficients (reflecting low water solubility). Air turbulence is expected to be moderate to high over most prairie potholes due to winds (see Sections 2.1 and 5.2.1).

Chemical and biological degradation processes can be influenced by pH, temperature and sunlight. Hydrolysis of herbicides in potholes depends primarily of the availability of hydrolyzable groups and secondly on the pH of the pond. Although pothole water from various areas may have a wide range of pH values, many potholes are neutral to slightly alkaline (see Section 2.2), a condition which facilitates hydrolysis.

Shallow pothole ponds are generally turbid and thus sunlight penetration is restricted. Direct photolysis, therefore, is expected to be an important degradation pathway only for that portion of the herbicide dequestered near the water surface. Sensitized photoreactions may be facilitated by the concentration of dissolved organic materials in some potholes.

Shallow ponds are detrital-based ecosystems with well developed microbial communities. During the warm period of the year, biodegradation rates for herbicides are expected to be reasonably high as rate expressions are generally assumed to be second order depending on the concentration of microbes.

7.1.2 Partitioning of herbicides in slough ecosystems

7.1.2.1 Physico-chemical information

Highly water-soluble herbicides (>10% weight/volume) were found at relatively high concentrations in surface runoff if applied mainly to foliage but not if applied to sparse vegetation or bare soil (Wauchope 1978). Once in the slough ecosystem some, but not all, highly water-soluble compounds are expected to remain in solution and to be bioavailable to plankton and submergent macrophytes. Because of their high water solubility values, strong acids of the aliphatic and aromatic acid groups are expected to be primarily found within the water column of a shallow slough (see Table 7.2). A low affinity for sediments is also suggested by the K_{OW} and K_{OC} values, Dalapon is among the most mobile of the pesticides (Helling 1971) and is not readily removed from water via adsorption (Foy 1975). On the basis of soil adsorption studies, dicamba and chloramben are expected to adsorb to sediments (Weber 1972). Loss of these acid herbicides from water via volatilization is unlikely at neutral pH where they exist as anions in solution. Negligible volatile loss was reported for dicamba from pond water (Scifres et al. 1973).

The amine and other salts of the phenoxy compounds are expected to dissociate in aqueous systems. As the dissociated forms are predominately anionic within a normal pH range (4-9), adsorption to negatively charged sediment particles is negligible (NRCC 1978b). Equilibrium sorption coefficients of 2,4-D acids and esters on clay minerals and on fulvic or humic-clay complexes were generally less than 100; ester forms were more strongly sorbed than the acids (Aly & Faust 1964, Khan 1974). Cochrane et al. (1967) reported a high degree of fenoprop PGBE ester in pond hydrosoils. The loss of phenoxy acids from water at neutral pH via volatilization is variable (Norris 1981). 2,4-D butyl and BEE ester were calculated to have half-lives of 1.1 and 895 days, respectively, based on volatilization losses (Zepp et al. 1975).

Large organic cations, such as the bipyridylium herbicides, are highly soluble but are rapidly adsorbed to negative charge sites on both clay and organic sediments and on plant surfaces (Bowmer 1982b). For example, diquat and paraquat were highly sorbtive to freshwater sediments and seston (Karickhoff & Brown 1978, Corwin & Farmer 1984). The high concentrations were found in the medium clay fraction and sorption maxima showed good correlation with cation exchange capacity (Karichkoff & Brown 1978). Little desorption of diquat at low concentrations (2,500 μg/g) observed. Volatile losses of the bipyridyliums are likely to be negligible due to their high water solubility and low vapour pressure.

Sorption of the carbamate herbicides by aquatic sediments has not been reported to date, however, strong adsorption of molinate to sediment was evident from its negligible volatilization loss from aqueous solutions in the presence of sediment (Deuel et al. 1978). K_{OC} values for triallate may approach 1,500 as predicted for thiobencarb (Muir 1986), indicating relatively strong sediment sorption potential. Volatilization of barban was estimated to proceed at a very slow rate (Muir 1985). Volatilization of this carbamate is a recognized loss pathway but fluxes were reduced by the presence of soil particulates (Deuel et al. 1978).

Table 7.2. Physico-chemical properties of selected herbicides and formulations.

Herbicide (Formulation)	Water Solubility at 25°C^a (mg L^{-1})	Vapour Pressure at 25°C^a (mm Hg)	K_{OW}^{b}	K_{OC}^{b}
Aliphatic and Aromatic Acids				
chloramben	700	6.98×10^{-3} (at 100°C)	13 [R]	12.8 [H], 189 [R]
dalapon	900,000	--	6 [K]	--
dicamba	4,500	3.41×10^{-5}	3 [R]	3.8 [H], 2.2 [R]
TCA	13,060,000	5 (at 77°C)	--	--
picloram	430	$6.16 \times 1-^{-7}$ (at 35°C)	2 [K]	6-24 [K], 12.7 [H], 26 [R]
Bipyridyliums				
diquat	700,000 (at 20%)	$<9.75 \times 10^{-8}$	--	--
paraquat	very high	$<1 \times 10^{-7}$	1 [R]	20,152 [H]
difenzoquat	765,000	--	--	--
Carbamates				
asulam	5,000	--	--	--
barban	11	3.79×10^{-7}	--	--
EPTC	375 (at 24°C)	3.4×10^{-2}	--	109 [K_n]
triallate	4	1.2×10^{-4}	--	--
Nitrites, Dinitroanilines, Amides				
propanil	225	9.1×10^{-3} (at 60°C)	106 [R]	--
propyzamide	15	8.48×10^{-5}	--	--
trifluralin	0.3	1.1×10^{-4}	9,328 [K_g], 218,776 [M], 1,150 [R]	-- --

Table 7.2 (cont'd)

Herbicide (Formulation)	Water Solubility at 25°C[a] (mg L^{-1})	Vapour Pressure at 25°C[a] (mm Hg)	K_{OW}[b]	K_{OC}[b]
Phenoxy Compounds				
2,4-D (acid)	900	6.0×10^{-7}	37 [K], 646 [V], 416 [R], 443 [R]	22-32 [K], 20 [R] 109 [R]
2,4-D (amine)	3,000,000 (at 20°C)	--	--	--
2,4-D (esters)	very low	(PGBE) 3×10^{-6}	--	--
dichloroprop (acid)	710	--	--	--
dichloroprop (Na salt)	660,000	--	--	--
MCPA (acid)	825	1.5×10^{-6} (at 21°C)	--	--
mecoprop	620 (at 20°C)	--	--	--
Triazines				
atrazine	33	5.7×10^{-8} (at 10°C)	562 [E], 212 [R]	102 [H], 163 [R]
cyanazine	171	1.6×10^{-9} (at 20°C)	--	--
simazine	3.5 (at 20°C)	6.1×10^{-9} (at 20°C)	88 [R]	159 [H], 138 [R]
metribuzin	1,220 (at 20°C)	$<1 \times 10^{-5}$ (at 20°C)	--	--
Urea Compounds				
linuron	75	1.5×10^{-5} (at 24°C)	~120 [E]	653 [H], 862 [R]
terbacil	710	4.8×10^{-7} (at 29.5°C)	78 [R]	41 [R]
Others				
amitrole	280,000	--	--	--
bentazon	500	$<0.1 \times 10^{-7}$ (at 20°C)	--	--
bromoxynil	130	--	--	--
diclofop-methyl	3,000 (at 22°C)	2.5×10^{-7} (at 20°C)	--	--
flamprop-methyl	35 (at 20°C)	3.5×10^{-7} (at 20°C)	--	--
glyphosate	12,000	--	.1 [G]	--

[a] Water solubility and vapour pressure data taken from Herbicide Handbook, 5th edition, 1983, and Pesticide Manual, 7th edition, 1984

[b] [K] - Kenaga 1980

[K_z] - Kanazawa 1981

[M] - Mackay 1982

[V] - Verschueren 1983

[E] - Ellgehausen _et_ _al_. 1980

[H] - Hamaker & Thompson 1972

[R] - Rao & Davidson 1982

[G] - Ghassemi _et_ _al_. 1981

[K_n] - Koren _et_ _al_. 1969

Equilibrium coefficients for atrazine and related chloro-s-triazines in natural stream and pond sediments are relatively low (Table 7.2) and increase with increasing organic matter content of the sediment (Brown & Flagg 1981). K_{OC} values for this group can be predicted adequately from K_{OW} values (Brown & Flagg 1981). K_{OC} values of $>100,000$ have been estimated for atrazine and natural colloids (<0.45 μm) (Means & Wijayaratne 1982). Volatilization loss of triazine herbicides is expected to be minimal due to their relatively low vapour pressures (Table 7.2) and Henry's constants of 10^{-8} to 10^{-10} (Muir 1986).

Low sorption to clay mineral and strong sorption to sediments and dissolved organic matter characterizes the phenyl urea herbicides (Muir 1986). Peck et al. (1980) reported a high correlation between the adsorption coefficients (Kp) of diuron on lake sediments and organic matter content. Linuron was strongly bound to colloidal matter (Means & Wijayaratne 1982). Losses of urea herbicides from water via volatilization is likely to be negligible. Vapour pressures are relatively low and solubility moderate to high (Table 7.2) providing low Henry's constants (Muir 1986).

Highly insoluble nonionizable compounds, such as the dinitroanilines and amides, sequester themselves in organic sediments and the lipophilic material of biota. Kanazawa (1981) reported a bioconcentration factor (BCF) of 3000 for trifluralin in fish. This BCF is substantially greater than that expected for most pesticides, with the exception of the highly bioaccumulated organochlorine compounds.

The fate of highly hydrophobic organic herbicides (compounds having water solubilities of less than a few mg L^{-1}) in natural waters is dependent upon their sorptive behaviour associated with nonpolar organic molecules and can be reasonably predicted by their partitioning coefficient with the organic carbon fraction of soil or sediment (K_{OC}) (Lambert 1967). This relationship is particularly pertinent to pothole pond sediments where fine organic particles in the sediment provide large K_{OC} values in comparison to sandy sediment and more heterogeneous soils. A number of authors have reported correlations between soil adsorption of hydrophobic pesticides and their octanol-water partitioning (e.g. Briggs 1973, Karickhoff et al. 1979, Brown & Flagg 1981). Generally, a good correlation between K_{OW} and K_{OC} is expected for potholes where nonspecific physical adsorption of neutral molecules is the predominant process. Conditions favoring sorbate dissociation, protonation or chemical interactions at the surface are expected to negate the assumption that organic materials are the primary sorbent.

Trifluralin had the highest K_{OC} value (30,550) of the selected herbicides listed in Table 7.2. Helling (1976) reported its strong adsorption to soils. The adsorption of amide herbicides to sediments has not been evaluated. Based on water solubility, propanil is expected to have a relatively low K_{OC} value and weak sorption behaviour; particulate sorption may be more important in the sequestering of propyzanide. Loss of dinitroanilines via volatilization from flooded soils is commonly observed. Brewer et al. (1982) found measurable losses of several dinitroanilines from flooded rice field soils. Losses of trifluralin to the atmosphere could be important in shallow ponds prior to the dominance of sediment sorption processes. No volatilization losses of the amide herbicide propanil from static water under rice culture was observed (Deuel et al. 1977).

Of the remaining unclassified herbicides in Table 7.2, amitrole is readily removed from soil by percolating water (Ghassemi et al. 1981), however its partitioning in pond water has not been examined. Anionic glyphosate is highly soluble but strongly adsorbed to the positive charge sites at the edges of clay minerals (Bowmer 1982a). Picloram is reported to be mobile in soils and not strongly adsorbed to soil organic matter (NRCC 1974).

It is obvious that the variety of chemical structures in herbicides makes prediction of partitioning based solely on physico-chemical properties difficult. The chemical nature of a herbicide also determines the influence of environmental variables such as pH on partitioning. Hamaker and Thompson (1972) developed a scheme relating chemical class to the impact of pH on soil adsorption. Table 7.3, modified from their scheme, shows examples for herbicides. The impact of pH on adsorption is expected to be greatest for herbicides which are weak acids and weak bases.

7.1.2.2 Laboratory and field studies

Herbicide compounds within the aliphatic and aromatic acid group (chloramben, dalapon, dicamba, TCA) showed the greatest mobility in the soil environment of 40 pesticides tested using soil thin-layer chromatography (Helling 1971). Therefore, these compounds are among those most likely to leach through or run off soils. Studies of the residues of dalapon and TCA, freely water-soluble herbicides used for weed control on ditchbanks, following application led researchers to conclude that these two compounds were not extensively sorbed from water (Frank et al. 1970). A second study examining the fate of dicamba also indicated that its distribution is largely unaffected by adsorption to sediments or plants (Scifres et al. 1973). Dicamba concentrations were not magnified in the aquatic food chain in studies with Metcalf microcosms (Yu et al. 1975b).

Adsorption of picloram is minimal in neutral to alkaline soils, but soil binding increases with an increase in soil organic matter content or decrease in soil pH (review in NRCC 1974). Kenaga (in NRCC 1974) reported that concentrations of residues in pond sediments immediately after spraying picloram were only twice those in water. After 75 days detectable residue levels were found in both water (0.1 μg L^{-1}) and sediment (7 μg kg^{-1}). After application of picloram (Tordon 10K) pellets to a watershed at a rate of 44.8 kg ha^{-1}, the highest residue levels of 437 μg L^{-1} in pond water and 657 μg/kg dry wt. in bottom sediment were found in a one- to three-week period (Dennis et al. 1977). The residue peaks following rainfall and sorption to suspended particulates was suggested as contributing to the high concentrations in unfiltered water samples. These data suggest that, unlike the other compounds in the acids group, picloram exposure must be evaluated in both the water and sediment compartment. Picloram is not concentrated from water or food in aquatic animals (NRCC 1974); thus, exposure is governed by residue concentrations in the medium.

Very little is known about the environmental distribution of carbamate herbicides in aquatic ecosystems. No pond studies were found in a search of the literature. The leaching of EPTC and diallate in soil columns was directly related to their water solubility and inversely related to the organic carbon content of the soil (Koren et al. 1969). These data suggest that compounds such as barban and triallate may be expected to partition into organic sediments of shallow ponds to a somewhat greater degree than EPTC. Carbamate herbicides were less strongly adsorbed to bentonite clays than urea compounds (El-Dib & Aly 1976b).

The distribution of diquat and paraquat in aquatic ecosystems has been widely studied, but there is little information on the environmental behaviour of difenzoquat. Diquat and paraquat dissipate rapidly (~4-12 days) in waters from natural aquatic systems (Grzenda et al. 1966, Yeo 1967, Calderbank 1970). The rapid disappearance of diquat and paraquat from water is attributable to their adsorption by sediment, suspended particulate matter and aquatic plants (Simsiman & Chesters 1976, Simsiman et al. 1976). Newman and Way (1966) reported that, within four days, aquatic macrophytes concentrated diquat by a factor of 50 over an initial concentration of 0.5 mgL^{-1}, suggesting that sorption of aquatic plants has a major influence on the distribution of the bipyridyliums in densely vegetated sloughs. As these compounds are potent contact herbicides, rapid plant death and decomposition carry that portion of the herbicide which is not degraded to the sediments. The partitioning of paraquat, diquat and difenzoquat to sediments and macrophytes is expected to be rapid with the availability of adsorbents controlling distribution and consequent exposure. Pond mud has been demonstrated to bind diquat reducing dramatically bioavailable concentrations and raising LC_{50} values for exposed amphipods by a factor of 100 (Wilson & Bond 1969).

Little is known about the distribution of nitriles, anilines and amides in aquatic ecosystems, although their physico-chemical properties suggest a tendency to accumulate in organic sediments and biota. Trifluralin was the only herbicide tested in the Metcalf microcosms that showed a propensity to accumulate in aquatic consumers (Metcalf & Sanborn 1975). Values for ecological magnification (EM) of trifluralin were 17,872 for the snail and 926 for the fish. These EM values are roughly equivalent to those for methoxychlor. Van Valin (1966) reported that dichlorobenil (a nitrile) in a farm pond treated to 0.6 mg L^{-1} had accumulated by day one to levels of 13 ppm in mud and over 500 ppm in filamentous algae.

Phenoxy herbicide compounds are mobile, particularly in marshy areas. Norris (1967) found 2,4-D movements to be well correlated with the raising of the water table. The NRCC (1978b) review of the phenoxy herbicide compounds states that they do not accumulate in the sinks (sediments, biotas) of aquatic ecosystems to the extent of the more hydrophobic pesticides. The actual amount of the phenoxy herbicide in the sediment is likely to be proportional to the organic content of bottom muds when the pH is low, but drastically reduced when the pH exceeds the pKa (see Table 7.3). Residues of 2,4-D (DMA) applied to a canal at 4.48 kg ha^{-1} reached a peak in water (0.04 mg L^{-1}) in one day and in hydrosoil (0.005 mg kg^{-1}) in 3-15 days (Schultz & Whitney 1974). A significant fraction of 2,4-D and MCPA were entrained in plants (Soderquist & Crosby 1975) and plankton (Wojtalik et al. 1971). Phytoplankton removal of 2,4-D is species and pH dependent with the greatest uptake in acid water (Valentine & Bingham

Table 7.3. Classes of compounds related to the effect of pH on adsorption of herbicides[a].

			Molecular form		
Class	Example	pK_a	Low pH	High pH	pH Effect
Strong acid	TCA		Anion	Anion	Small
Weak acid	Picloram	3.7[b]	Free acid	Anion	Large adsorption below pK_a and then decrease
Strong base	Diquat		Cation	Cation	Decrease at very low pH
Weak base	Ametryne		Cation	Free base	Increasing adsorption to pH approx. pK_a and then decrease
Polar molecules	Linuron		Un-ionized	Un-ionized	Small
Neutral molecules	DDT	Nil	Un-ionized	Un-ionized	Probably none

[a] Modified from Hamaker & Thompson 1972

[b] pK_a values for other weak acids: 2,4-D = 2.6-3.3, MCPA = 2.9-3.4, mecoprop = 3.1-3.2, dichlorprop = 2.9-3.0 (from NRCC 1978)

1974). Bioaccumulation of phenoxy herbicides is not a general phenomena (Metcalf & Sanborn 1975, Virtanen et al. 1979). Residues in phytoplankton and macrophytes persisted for 2-6 months (Wojtalik et al. 1971), suggesting that detectable residues may also be expected in aquatic grazers. Certainly, the non-exclusive partitioning of phenoxy compounds in aquatic systems suggests the potential exposure of pelagic, benthic and macrophytic species must be considered.

The mobility of triazines in soil is greater than EPTC and trifluralin but far less than the aliphatic acids. Helling (1971) concluded that triazines are rather immobile in soil. The amount of simazine in the mud and water of treated ponds was directly proportional to application rates (Mauck et al. 1976). Residues were slightly lower in water than mud, seldom exceeding 30-50% of the total amount applied. The introduction of sediment to laboratory systems more than doubles the loss of simazine from water (Tucker & Boyd 1981). Klaassen and Kadoum (1979) found nearly equal concentrations of atrazine in water, mud and biota one day after treatment of a farm pond at 0.3 mg L^{-1}. The rate of decay with time was greatest in the surface water. Dilution of treated water brought triazine out of the sediment and into the soluble phase thereby maintaining toxic levels (Goldsborough & Robinson 1983). Surprisingly, triazine residues were found in most water samples from streams feeding the Great Lakes, but in none of the suspended particulate samples (Frank & Sirons 1979). Cyanazine (Yu et al. 1975a) and simazine (Hawxby & Mehta 1979) accumulated to substantial concentrations in algae (10-20% of applied). Atrazine removal from culture water by algae was rapid (<6 h) and unaffected by light and temperature (Valentine & Bingham 1976); atrazine was not metabolized by algae (Butler et al. 1975a, Valentine & Bingham 1976). Because of their high toxicity to algae, the triazines may be rapidly turned over in the water column. These compounds have been shown to accumulate in invertebrates and fish (Mauck et al. 1976, Klaassen & Kadoum 1979) but are apparently not concentrated in the food chain (Yu et al. 1975a).

Linuron and other urea-based compounds were relatively tightly bound by the organic carbon fraction of soil and sediment (Hance 1965, Hamaker & Thompson 1972, Peck et al. 1980). The partitioning behaviour of linuron is expected to be similar in soil and sediments with equivalent organic content (Peck et al. 1980). Adsorption of the urea compounds to clay particles was less important in determining their distribution in aquatic systems (El-Dib & Aly 1976b). Data on surface water residues in runoff from agricultural fields suggest that desorption is minimal. The urea compounds are less tightly held than trifluralin but are ten times more immobile than an aliphatic acid like chlorfenac (Willis et al. 1975).

7.1.2.3 Partitioning summary

There is surprisingly little data on the environmental behaviour of herbicides in aquatic ecosystems. Much of the data suggesting distribution favoring the sediment compartment comes from soil adsorption studies. As prairie sloughs generally have sediments rich in organic matter and relatively dense macrophyte vegetation, adsorbent surfaces may be more available than in agricultural soils. Based on soil and aquatic partitioning studies and inferences from structurally similar compounds, the expected equilibrated distribution of selected herbicides in contaminated sloughs is summarized in Table 7.4. Some uncertainty is attached to these classifications due to the general lack of data. The strong and weak acids are expected to remain in water with strong adsorption of weak acids only below pH values equal to pK_a. The triazine compounds are expected to be divided between water, sedimets and macrophytes. Nonionizable hydrophobic compounds like linuron and trifluralin will partition almost exclusively to sediments, macrophytes and algae. The less soluble carbamates like triallate are also expected to favor organic materials.

7.1.3 Persistence

The persistence of herbicides in slough ecosystems will be determined primarily by the rate of chemical, photochemical and biological degredation. Microbiological transformations are, in most cases, a major pathway for herbicide degradation and microbial activity is the most important (but highly unpredictable) rate-controlling factor. Soil type, water content, pH, temperature and clay and organic matter content also influence the rate of degradation of herbicides. In flooded soils or deep slough sediments, herbicide degradation rates will depend on the relative activity of aerobic versus anaerobic microbes (review by Williams 1977).

7.1.3.1 Less persistent herbicides

The phenoxy herbicides have been well studied and have demonstrated little persistence in sediment or water. They are primarily degraded by biological processes, although the hydrolysis of phenoxy alkanoic esters to their acids in solution and soil, as well as the dissociation of amine salts in water, are important chemical mechanisms. Hydrolysis of the ester is rapid, particularly in alkaline conditions. Zepp et al. (1975) reported the hydrolytic half-life of various 2,4-D esters in water to be from 1.5-37 hours at pH 9 but from 44-1,500 days at pH 6. Under field conditions, the hydrolysis of phenoxy alkanoic esters to their respective acids in soils should occur within 2-3 days at normal moistures and at temperatures above the melting point (NRCC 1978b). The DMA salt of 2,4-D undergoes dissociation in water and moist soils (Grover & Smith 1974). The product of the hydrolysis of the ester or of the dissociation of the amine salts is the same phenoxy alkanoic amine. Photochemical degradation of phenoxy herbicides has not been demonstrated in natural aquatic systems and should be considered insignificant compared with biological degradation (NRCC 1978b). Biodegradation in soils is favored by moist, warm conditions and high organic matter. In water and sediments, aerobic conditions are required for significant microbial degradation (consult review in Loos 1975; NRCC 1978b). Microbial activity in the breakdown of 2,4-D in aquatic systems was related to nutrient content, temperature and the distribution of sediment (Nesbitt & Watson 1980a, b) as well as to the size and nature of the microbial populations (Hemmett & Faust 1969).

Field studies from shallow ponds showed that 2,4-D degraded from 4.5 mg L^{-1} to <0.05 mg L^{-1} in 14 days (Robson 1966). In treatments of 0.1 to 5 mg L^{-1}, 2,4-D residues in pond water were only 10-30% of initial level

Table 7.4. Expected equilibrated distribution of selected herbicides in a contaminated slough ecosystem.

Primarily in Water	Divided Between Water and Sediment-Macrophyte	Primarily in Sediment-Macrophyte
amitrole	asulam	barban
2,4-D (acid, salts)	atrazine	difenzoquat
dalapon	chloramben	diquat
dicamba	cyanazine	EPTC
dichlorprop	glyphosate	linuron
MCPA	metribuzin	paraquat
mecoprop	picloram	triallate[a]
TCA	simazine	trifluralin[a]
	2,4-D esters[a]	

[a] Potential volatilization losses

after 20 days (Cope et al. 1970). Aly and Faust (1964) reported that 2,4-D at 3 mg L^{-1} persisted substantially longer in lake water without sediment (>120 days) than in water with sediment. The herbicide disappeared from lake mud in 65 days after an initial treatment and 85% disappeared within one day with repeated additions to a previously exposed sediment microbial community. MCPA disappeared from rice pond water and sediment within a two-week period following an application of 0.98 kg ha^{-1} (Soderquist & Crosby 1975).

The strong acid herbicides are relatively rapidly degraded. In impoundment water treated at an initial concentration of 11 mg L^{-1} dicamba, dissipation was complete after about 40 days (Scifres et al. 1973). Light and sediment increased the loss rate of dicamba. Yu et al. (1975b) reported that dicamba persisted in the water of a Metcalf microcosm in conjugated and anionic forms and that only 10% was transformed in 30 days. Dalapon is expected to persist in water for two to three days and in sediment for up to 60 days (Newbold 1975).

7.1.3.2 More persistent herbicides

Adsorption to clays has been shown to modify the availability of herbicides to microbial degradation. For example, montmorillonite-adsorbed diquat was protected from microbial attack (Weber & Coble 1968). If diquat occurs in soils and aqueous solutions, photochemical degradation may compete with microbial degradation, but all processes are slow for the adsorbed compound (Simsiman & Chesters 1976). Diquat adsorbed to lake sediment persisted intact for more than 180 days: less than 2% was broken down in three months (Ibid). The rate of degradation was more rapid for diquat associated with macrophytes. Following weedkill, a profusion of microorganisms occurred, promoting the degradation of the diquat sorbed on the decomposing macrophytes. Within 22 days, 32% of the herbicide had been degraded. Way et al. (1971) found that residues of paraquat in sediment from a small lake decreased slightly between 25 to 60 weeks following treatment, but were relatively stable between 86 to 110 weeks (11 mg kg^{-1} dry weight of mud). Newbold (1975) suggested that diquat and paraquat can persist in mud for >1 year and >2 years, respectively.

Urea herbicides can be degraded through biochemical, chemical and photochemical mechanisms (review Maier-Bode & Hartel 1981). Chemical hydrolysis was only effective in highly alkaline solutions (El-Dib & Aly 1976a). Mixed populations of microorganisms from river water did not metabolize linuron in a four-month test period. Enriching the microbial inoculum causes greater than 60% of the herbicide to be degraded in two weeks (El-Dib & Aly 1976c). The loss of linuron from enclosures in ponds was adequately described by an exponential regression with the half-life at 16 to 40 days (Stephenson & Kane 1984). Diuron was completely degraded in enriched cultures of anaerobic sediment in 17 to 25 days; residues in the sediment were 300 ng g^{-1} at three weeks following treatment and had dissipated to 31 ng g^{-1} in eight months (Attaway et al. 1982a, b).

The triazine compounds are of interest because they distribute relatively equally between water and sediment and are moderately persistent in each compartment. The half-life for atrazine may exceed three months in both sediment and water systems (Geller 1980). Simazine had a half-life of >32 days in pond water without sediment and 12 days in the presence of sediment (Tucker & Boyd 1981). Microbial degradation is a significant factor affecting the residual life of triazine herbicides in the environment. In contrast to phenoxy compounds (and other herbicides) where the microbial community may require a lag period to build up adapted populations, there was little evidence of a lag phase in the degradation of triazines, although the continuous rate of degradation was slow (Kaufman & Kearney 1970). A build-up of effective microbial degraders did not occur with repeated applications of most triazines. Anaerobic conditions limited degradation (Skipper _et al_. 1967). An increased triazine degradation rate at high temperature may be a combination of increased microbial activity coupled with increased solubility and availability of the compounds (Kaufman & Kearney 1970).

The first major reaction in the degradation of the triazine may be chemical rather than biological. Chemical hydrolysis of the triazine molecule occurs under field conditions and is probably favored at both high and low pH conditions. Degradation in soil was most rapid at pH >5.0 (Kells _et al_. 1980). Neutral pH may favor a limited bacterial attack (Kaufman & Kearney 1970). Atrazine does not appear to be metabolized by algae (Valentine & Bingham 1976).

In small ponds, simazine applied at treatments of 0.1 to 3.0 mg L^{-1} persisted in sediment and water at detectable levels (>0.001 µg g^{-1} and >.01 mg L^{-1}, respectively) for more than one year (Mauck _et al_. 1976). The estimated half-life of simazine was 7-30 days in pond water and 15-45 days in the benthic sediments in this study. Simazine concentrations in the water were possibly influenced by wind mixing of bottom sediments into the water. Smith _et al_. (1975) reported that simazine and atrazine were more persistent in irrigation ditches than two urea herbicides. In another study, atrazine residues remained in pond water at 206 µg L^{-1}, 120 days after a 300 µg L^{-1} application (Klaassen & Kadoum 1979). Approximately 300 ppb remained in benthic mud at the end of the same period.

Trifluralin displayed a tendency towards moderate persistence in anaerobic sediments, approaching a 20-day half-life (Isensee _et al_. 1979). This dinitroaniline herbicide can be degraded both photochemically and biologically (Parr & Smith 1973). Unlike the triazines and phenoxy compounds which are not degraded anaerobically, trifluralin's biological degradation was more rapid under anaerobic than aerobic conditions (Parr & Smith 1973). The half-life for aerobic degradation of trifluralin in soils was >1 year (Messersmith _et al_. 1971), but in an aquatic model ecosystem, 60% of trifluralin was degraded in flooded anaerobic soil by day 70. This herbicide was more than 90% degraded in microcosm water within nine days (Isensee _et al_. 1979).

There are few published reports on the persistence of thiocarbamates in natural waters. Slow degradation of molinate has been observed in flooded rice culture soils; a half-life of about 16 days was estimated (Deuel _et al_. 1978). Soderquist _et al_. (1977) found that molinate disappeared rapidly from water (half-life <3 days) following aerial application to a flooded rice field. Only a small portion of molinate partitioned to flood soils (~4%) but this residue persisted at >50% of its initial concentration for 100 days. Hydrolysis and photolysis are thought to be negligible contributors to the degradation process (Muir 1986).

Picloram residues in pond sediment persisted for up to 75 days (7µg kg^{-1} compared to an initial 148 µg kg^{-1}). Levels in water were 0.1 µg L^{-1} at day 75, indicating a slow rate of biolysis (from NRCC 1974). Picloram is stable to hydrolysis (Weidner 1974) but can be photodegraded (Hedlund & Youngson 1972).

Glyphosate was found to be 90% dissipated within 30 days in water covering soils treated at 4.4 kg ha^{-1} (Sacher 1978) but had a longer half-life in other natural waters: seven weeks in a sphagnum bog and nine weeks in a cattail swamp (USEPA data in Ghassemi _et al_. 1981). A single report by Van der Weij (1966) suggested that amitrole was highly persistent in microcosm water (half-life -300 days).

Based on the very limited data available for aquatic persistence of the major use herbicides, a classification scheme is suggested as a basis for chronic exposure assessment (Table 7.5). Based on known persistence in aquatic sediments, diquat, paraquat and linuron should be assessed for potential chronic effects. Under aerobic conditions trifluralin is highly persistent. Trifluralin as well as triallate and picloram are classified as persistent because of the documented residue carry-over (>25% in treated fields; see Table 7.1). Volatilization may decrease the persistence of trifluralin and triallate in potholes. The triazines, although listed as only moderately persistent, should be considered a potential chronic threat to slough biota because of the high degree of availability in water, the evidence of a lack of adaptability of microbes and the relatively high toxicity to plants.

Table 7.6 presents a summary of field data on herbicide persistence in soil, pond sediment and water. Data for the half-life of these herbicides in soil was taken from a review by Rao and Davidson (1982). The bipyridyliums are highly resistent to microbial attack when tightly bound to soil, clay or organic matter. The urea and uracil compounds are also persistent in soils followed, in decreasing order, by the triazines > glyphosate, trifluralin and picloram > aliphatic and aromatic acids and phenoxy compounds.

In summary, field dissipation of phenoxy acids is relatively rapid and is likely due to biolysis. Previous exposure of degraders to the phenoxy acids increases the rate of breakdown. Persistence in sediment of >30 days has been observed in macrophyte systems but is not expected for productive potholes. Volatile losses of phenoxy esters are expected as is their persistence in sediment. Aliphatic and aromatic acid herbicides are expected to dissipate rapidly. Triazine compounds are expected to degrade moderately slowly via hydrolysis and biolysis. The bipyridyliums dissipate rapidly from water but very slowly from sediment where they sequester. Dinitroaniline urea and amide compounds degrade relatively slowly when sequestered in sediments. Biodegradation of the thiocarbamates is expected to be slow but volatilization losses may be important in shallow ponds under turbulent prairie atmospheric conditions.

Table 7.5. Grouping of selected herbicides according to expected persistence in pond ecosystems.

Non-Persistent $t_{1/2} < 20$ days	Moderately Persistent $20 \text{ days} < t_{1/2} < 100$ days	Persistent $t_{1/2} > 100$ days
2,4-D	atrazine	diquat
MCPA	simazine	paraquat
dichlorprop[a]	cyanazine	difenzoquat
mecoprop[a]	metribuzin[a]	linuron
dicamba	glyphosate	terbacil
dalapon	EPTC[a]	EPTC[a]
TCA	barban[a]	triallate[abc]
	asulam[a]	trifluralin (aerobic conditions)[c]
	propanil[a]	
	trifluralin (anaerobic conditions)[c]	picloram[b]

[a] Group selected on basis of data from similar compounds

[b] Based on soil persistence data from Smith (1982)

[c] Volatilization losses may decrease persistence

Table 7.6. Persistence of selected herbicides under field conditions

Herbicide	Soil Half-Life^a (days^{-1})	Pond Sediment Half-Life^b (days^{-1})	Pond Water Half-Life^b (days^{-1})
Phenoxy Compound			
2,4-D	<1-15	2-5 [C]	7-14 [C]
Triazines			
atrazine	12-53	>120 [K]	>120 [K]
simazine	11-246	15-45 [M]	7-30 [M]
Bipyridyliums			
diquat	--	~2,250 [S]	1-4 [S] [Y]
paraquat	4,747	--	1-4 [G]
Dinitroanaline			
trifluralin	15-69	~20 [I]	~2[I]
Urea Compounds			
linuron	230	~90-120 [A]	16-40 [SK]
terbacil	175	--	--
Aliphatic and Aromatic Acids			
dicamba	6-10	--	4 [SA]
dalapon	15	--	--
picloram	12-84	--	--
TCA	15	--	--
Others			
glyphosate	3-130	--	12 [R]
amitrole	--	--	300 [W]

Table 7.6 (cont'd)

[a] Soil half-life data taken from Rao and Davidson 1982

[b] [C] - Cope _et al_. 1970

[K] - Klaassen and Kadoum 1979

[M] - Mauck _et al_. 1976

[S] - Simsiman and Chesters 1976

[Y] - Yeo 1967

[G] - Grzenda _et al_. 1966

[I] - Isensee _et al_. 1979; microcosm study

[A] - Attaway _et al_. 1982a; for diuron

[SK] - Stephenson and Kane 1984

[SA] - Scifres _et al_. 1973

[R] - Sacher 1978

[W] - Van der Weij 1966

7.1.4 Exposure analysis

Initially, toxicity data for all herbicides will be compared to the maximum expected concentrations of the herbicide assuming 100% transfer of the compound to pond water or margin from aerial overspray at field recommended application rates (AR). Table 7.7 summarizes calculated concentrations under the conditions described above for a shallow pond model as applied to the hazard analysis in Section 6.3. Only dalapon applied at >12 kg ha^{-1} leads to an expected concentration in water of greater than 1 mg L^{-1}. Amitrole, dicamba, EPTC, glyphosate and simazine might hypothetically approach 0.5 mg L^{-1}. Under our initial assumption, most of the other herbicides would be present at levels from 0.1-0.5 mg L^{-1}. Only those herbicides with demonstrated toxicity at or below the calculated rate will be considered an acute threat. For these compounds, more realistic exposure scenarios will be developed accounting for 1) mode of application (aerial vs. ground); 2) average on-target deposit from a single swath (~50% AR for aerial sprays) or from multiple swaths (~80% AR for aerial sprays); 3) maximum expected input from short distance off-target drift (~10% AR); 4) maximum expected runoff ($<$1-20% AR); and 5) maximum expected carry-over in sediments. The contribution via runoff will be compound-specific and based on the field data discussed in Section 5.3.3. Maximum sediment residue from carry-over will be based on Table 7.1 and Figure 7.1.

Impacts stemming from chronic exposure will be examined only for those compounds listed as persistent in Table 7.6 and for the triazine compounds for reasons previously stated.

7.2 Herbicide Effects on Vegetation

The evaluation of the impact of over 30 herbicides on prairie potholes is a large task which can only be done by sacraficing detail. Both the lack of a complete understanding of prairie pothole dynamics and the paucity of experimental ecotoxicological work on herbicides in aquatic systems restricts this evaluation. In addition, the 31 herbicides used in the Canadian prairies belong to more than seven different categories which differ in their mode of action, use, toxicity and physico-chemical properties.

Because of these roadblocks, the evaluation of herbicide impacts must proceed through the extrapolation to field situations of direct toxicity work. The predictions can be checked against experimental studies done in the field field which, unfortunately, consist mostly of studies aimed at evaluating the aquatic weed control potential of certain herbicides.

This evaluation of herbicide impact on prairie potholes begins with a focus on their effect on the vegetation because plants are clearly the dominant component of sloughs with respect to both biomass and system dynamics (see Chapter 2).

Table 7.7. Normal application rates and expected initial concentrations in pond water from aerial overspray of herbicides.

Herbicide	Application Method[a]	Recommended Application Rate[b] (g AI ha^{-1})	Expected Initial Concentration[c] in Pond Water (µg L^{-1})
amitrole	G	7,000	860
asulam	G	1,100	135
atrazine	G	3,360	413
barban	A, G	420	52
bentazon	G	1,080	133
bromoxynil	A, G	340	41
chloramben	G	2,220	266
cyanazine	G	2,400	295
2,4-D - low volatility ester	A, G	912-1,600 (forage)	92-197 (forage)
- amine		900-2,125 (forage)	108-261 (forage)
dalapon	A, G	7,295-12,580 (forage)	155-1,546 (forage)
dicamba	G	280-4,400 (forage)	34-541 (forage)
dichlorprop	G	525-3,850 (brush)	65-473 (brush)
diclofop-methyl	A, G	713	88
difenzoquat	A, G	840	103
diquat	A, G	550	68
EPTC	G	4,400	541
flamprop-methyl	A, G	504	62
glyphosate	G	1,246-4,272 (perennial weeds/brush)	153-525 (perennial weeds/brush)
linuron	G	264-2,160 (vegetable/shelter belt)	32-265 (vegetable/shelter belt)
MCPA - K salt	A, G	800	98
- Na salt		825	101
- amine		875	108
- ester		875	108
MCPB	A, G	1,700	209
mecoprop	G	1,050	129

Table 7.7 (cont'd)

Herbicide	Application Method[a]	Recommended Application Rate[b] (g AI ha^{-1})	Expected Initial Concentration[c] in Pond Water (μg L^{-1})
metribuzin	G	264	32
paraquat	G	1,100	135
picloram	G	24-1,500 (brush)	3-184 (brush)
propanil	G	990	122
propyzamide	G	1,625 (forage)	200 (forage)
simazine	G	2,150-6,450 (shelter belt)	264-793 (shelter belt)
TCA	G	3,735	459
terbacil	G	1,200 (forage)	147 (forage)
triallate	A, G	2,200	270
trifluralin	G	1,635	201

[a] G = ground rig; A = aerial spray

[b] Rates based on highest application levels recommended for cereal grain or oil seed crops as taken from agricultural pamphlets from Alberta, Manitoba and Saskatchewan

[c] Expected concentration estimated for direct overspray of model pond with surface area = 4,550 m^2 and volume = 3.7×10^6 L

7.2.1 Direct toxicity

Wetland plants consist of two types: the algae and the vasuular plants. These differ significantly in their dynamics and in their roles within the potholes. The algae have generation times in the order of days, can competitively displace other algal species very rapidly and their biomass is recycled through consumers or decomposition much faster than is macrophyte biomass. Therefore, not only are there differences in their responses to and recovery from exposure to herbicides, but the indirect consequences of such impacts should also be different. In light of this, the direct toxicity of herbicides to algae and vascular plants will be examined separately.

7.2.1.1 Algae

Studies on the toxicity of herbicides to algae consist mainly of in vitro work done on one species or an assemblage of species in liquid media culture. The focus is put on one or more of the following variables of interest: chlorophyl a levels, biovolume, photosynthesis as measured through the assimilation of radio-labelled bicarbonate, growth rate or yield and, rarely, respiration. Table 7.8 summarizes some of the work done on herbicide toxicity to algae. The values reported are the ranges of herbicide concentrations over which a significant inhibitory effect was observed on at least one of the variables listed above. Most of these concentrations consist of EC_{50}'s. The lowest values reported were used as an index of the relative toxicity of the 21 compounds, which are listed in decreasing order of toxicity.

Although the toxicity studies dealt with a large variety of algal species (eg. unicellular greens, soil algae and filamentous epiphytes), the responses of which differed substantially for any given herbicide, the ranking fairly accurately reflects the phytotoxic potential of these herbicides. Eight of the 21 substances listed are toxic to some algae at levels of 0.1 mg L^{-1} or below: three triazines, two carbamates, two bipyridyliums and one aromatic acid. The top four herbicides, along with paraquat, dicamba and dalapon, are of particular interest because they would be found in pond waters after overspray at concentrations exceeding levels at which they are toxic to algae (Table 7.8). The two triazine herbicides, atrazine and simazine, are clearly the most toxic and the greatest source of concern as the expected initial concentration in pond water is more than two orders of magnitude greater than levels causing inhibition.

A factor which may increase the potential impact of these compounds on algae is persistence. Compounds with long half-lives in aquatic systems may build up to toxic levels if used frequently enough. This may be the case for paraquat, diquat, picloram, triallate and glyphosate which can be highly persistent under certain conditions. Although the last three are toxic only at levels of 1 mg L^{-1} or greater, they have the potential to accumulate to such levels in pond water (see Section 7.1). Triallate is one of the most heavily used herbicides in the Canadian prairies.

Table 7.8. Herbicide concentrations tested which resulted in partial or total inhibition of growth or photosynthesis in various algal species. Also listed are the expected concentrations of herbicides in pond water immediately following overspray at recommended application rates.

Herbicide	Herbicide Concentrations Tested (ppm)	Expected Initial Concentration in Pond Water in ppm[a]	Sources[b]
atrazine	0.001-5.0	0.413	3, 6, 7, 9, 10, 11
simazine	0.001-1.0	0.264-0.793	6, 7, 8, 11
linuron	0.025-10.0	0.032-0.265	5, 6, 7
EPTC	0.025-125.0	0.541	6
metribuzin	0.05-1.0	0.032	1
diquat	0.1-10.0	0.068	5, 12
paraquat	0.1-25.0	0.135	5, 6, 12
barban	0.1-25.0	0.052	5, 6, 7
dicamba	0.1-0.5	0.034-0.541	5
propanil	0.18-1.0	0.122	7, 13
dalapon	0.2-75.0	0.155-1.55	5, 6
2,4-D	0.5-250.0	0.09-0.26	3, 5, 6, 11, 12
bromoxynil	0.5-10.0	0.041	5, 7
picloram	1.0-7.0	0.003-0.184	5
terbacil	1.0	0.147	7
TCA	2.5-125.0	0.459	6
triallate	2.5-125.0	0.270	6
glyphosate	3.0	0.153-0.525	4
asulam	6.0	0.135	13
MCPA	75.0-250.0	0.1	6
amitrole T	125.0-250.0	0.86	6

[a] From Table 7.7

[b] (1) Arvik *et al.* 1973; (2) Bednarz 1981; (3) Butler *et al.* 1975b; (4) Christy *et al.* 1981; (5) Cullimore 1975; (6) Girman 1976; (7) Kratky and Warren 1971; (8) O'Neal and Lembi 1983; (9) Stratton 1984; (10) Stratton *et al.* 1980; (11) Torres and O'Flaherty 1976; (12) Ware and Roan 1970; (13) Wright 1972

Extrapolation from this data to a field situation is difficult because of the variability of the response of algae to herbicides and the complexity of plankton communities. The range of concentrations inhibiting algal processes is wide for each compound. Bednarz (1981) showed that concentrations of 2,4-D needed to inhibit growth in algae ranged from 0.0001 ppm for Chlorella pyrenoidosa to 100 ppm for Oscillatoria sp. The range may also be wide for species within the same genus: the EC_{50}'s of simazine on the growth rate of two species of blue greens were 0.1 and 5.0 ppm for *Anabaena inaequalis* source and *Anabaena variabilis*, respectively (Bednarz 1981). In addition, low levels of certain herbicides stimulate growth. Of 12 algal species tested, six showed increased growth when exposed to concentrations of 2,4-D ranging from 0.01 ppm to 10.0 ppm (Ibid). Atrazine and simazine inhibit chlorophyll production in *Chlorella vulgaris* at 0.001 mg L^{-1} but stimulate it at concentrations of 0.0001 mg L^{-1} and 0.0005 mg L^{-1}, respectively (Torres & O'Flaherty 1976).

The responses of multi-species systems are more complex. In her study of algal responses to 2,4-D, Bednarz (1981) showed not only that the herbicide modified competitive interactions in mixed cultures by allowing more resistant species to prevail, but also that the latter decreased the toxicity of 2,4-D to the more sensitive species. This effect was clearly demonstrated in the treatment of ponds with atrazine where, at a concentation of 0.5 mg L^{-1}, phytoplankton succession was modified by the appearance and subsequent dominance of resistant strains of certain species (deNoyelles *et al*. 1982). Previous exposure to herbicides does not necessarily lead to increased resistance to subsequent inputs. The chronic exposure of artificial stream periphyton communities to 0.01 mg L^{-1} of atrazine, MSMA or paraquat did not modify the inhibition of the algae by these herbicides at concentrations of 1 or 10 mg L^{-1} (Kosinski & Merkle 1984). In addition, this study revealed that artificial stream algal communities derived from streams draining farmland, where herbicide use is heavy, were severely inhibited by 10 mg L^{-1} of paraquat whereas the same treatment had little effect on communities derived from unspoiled springs.

Clearly, field studies on pond ecosystems are necessary in order to assess the impact of herbicides on the algal component of prairie sloughs. Work done on multi-species algal systems in the field (deNoyelles *et al*. 1982, Goldsborough & Robinson 1983, Kosinski & Merkle 1984) showed that the concentrations of herbicide needed to produce a significant effect are 10 to 1,000 times greater than the lowest values recorded for single-species studies (Table 7.8), thereby suggesting that the potential for impact may be small. In addition, these field studies revealed that the recovery time of these systems ranged from 1 to 3 weeks. Therefore, acute exposure to the more toxic herbicides such as the triazines may reduce the productivity of the algal compartment at high levels of input but not for more than three weeks. Of greater concern, mostly because of lack of information, is the effect of chronic levels of those herbicides which carry-over from one year to the next, thereby increasing residual concentrations.

7.2.1.2 Macrophytes

The following section deals with the toxicity of herbicides to the dominant component of wetland vegetation: the macrophytes. These consist of all vascular plants found within the vegetation zones bounded by the wet meadow zone as well as macrophytic alga, Chara. Consideration will be given to plant life form as emergent, submergent and floating vegetation differ in their mode of exposure to herbicide application. The focus will be put on direct toxic effects to the plants. The indirect effects on plant community interactions will be dealt with separately.

Almost all of the studies reviewed were concerned with herbicide impact on aquatic weeds as target species. Because these efficacy studies were carried out in a variety of laboratory and field situations, the format and detail of the impact assessment is highly variable. In addition, the objectives of these studies do not concur with our attempts to evaluate the impact of herbicide use on wetland vegetation as non-target species.

The available information is presented in two parts because the studies are of two general types. This division is based primarily on the form of herbicide application and the reason for which it was applied. First, those studies monitoring the effect of aerial application of herbicides on target vegetation, mostly emergents, in marshes, lakes, canals or other bodies of water will be discusssed (Table 7.9). Second, the work done on the monitoring of the effects of herbicides on vegetation, mostly submergents, where mortality can be related directly to concentrations in the water of the compound used is summarized (Table 7.12). These studies consist mostly of experimental work done in the laboratory or in ponds.

Examination of Table 7.9 shows the difficulty involved in comparing the efficacy of three herbicides. First, concentrations of the compounds in the water were not reported, possibly because the water concentrations may not be relevant when the targets are emergents which pick up the herbicide through the aerial parts of the plants. Second, the habitats, species compositions and target species differed between studies. However, interspecies comparisons may still be made for a given compound: the percentage of plants killed in a marsh by application of the butyl ester of 2,4-D varied between 0 for Phragmites communis and 100 for Polygonum coccineum (Hanson 1952). A third difficulty involved in the ranking of the compounds with respect to their phytotoxicity is the wide range of application rates: 2,4-D amine was applied at a rate of 0.56 kg a.i. ha^{-1} to control emergents in a marsh (Hanson 1952) while the butoxyethanol ester of 2,4-D was sprayed at a rate of 112 kg a.i. ha^{-1} to eliminate submergents in an impoundment. If one judges toxicity by the amount of pesticide required to control the vegetation, then Table 7.9 indicates that glyphosate is possibly the most toxic.

Extrapolation to the situation in the Canadian prairies is not possible due to differences in the rates of application required to control crop weeds and aquatic weeds (Table 7.9). This is more clearly illustrated in Table 7.10 which lists recommended rates for crop weed control and rates for aquatic weed control: except for glyphosate, more herbicide is usually recommended for the elimination of macrophytes. This may be due to the greater "coarseness" of the foliage of aquatic vegetation or because, in

Table 7.9. Effect of aerial application of three herbicides on aquatic vascular plants.

Herbicide	Ecosystem	Aerial Application Rates (kg a.i. ha^{-1})		Macrophyte Species ª and Control	Source
		Rate used in study	Recommended rate for use in Prairies		
2,4-D amine	North Dakota marsh	0.56	0.9-2.1	Polygonum coccineum (E) 100% killed Eleocharis palustris (E) 50% " Scirpus paludosus (E) 20% " S. acutus (E) 60% "	Hanson 1952
2,4-D butyl ester	--	1.12	0.9-1.6 (for low volatile esters)	Polygonum coccineum (E) 100% killed Eleocharis palustris (E) 70% " Scirpus paludosus (E) 90% " S. acutus (E) 40% " S. americanus (E) 50% " Phragmites communis (E) 0% "	Hanson 1952
2,4-D butoxyethanol ester	coastal impoundment	112.0	0.9-1.6 (for low volatile esters)	Myriophyllum spicatum (S) Ruppia maritima (S) Najas guadalupensis (S) Potamogeton peutinatus (S) eliminated or greatly reduced	Getsinger et al. 1982
2,4-D dimethylamine	reservoir	22.4	0.9-2.1	Myriophyllum spicatum (S) 100% killed	Wojtalik et al. 1971
dalapon	canals	21.25	1.3-12.6	Typha angustifolia (E) > 55% reduction Typha latifolia (E) > 50% " Phragmites communis (E) > 65% " Juncus effusus (E) > 55% "	Stankovic et al. 1982
glyphosate	lake	1.0	N/A	Nuphar lutea (Fl) 98% killed " 99.9% "	Barrett 1974

Table 7.9 (cont'd)

Herbicide	Ecosystem	Aerial Application Rates (kg a.i. ha^{-1})		Macrophyte Species[a] and Control		Source
		Rate used in study	Recommended rate for use in Prairies			
glyphosate (cont'd)	review of efficacy against aquatic weeds	1.8	N/A	*Alisma* spp. (E)	> 70% control	Evans 1978
		1.8		*Carex* spp. (E)		
		2.0		*Eleocharis* (E)		
		1.44		*Glyceria maxima* (E)		
		1.8-2.88		*Juncus effusus* (E)		
		1.44		*Lemna minor* (Fl)		
		1.8		*Phalaris arundinacea* (E)		
		1.8		*Phragmites communis* (E)		
		1.44		*Rorippa nasturtium aquaticum* (E)		
		2.0		*Salix* spp. (E)		
		2.66		*Scirpus lacustris* (E)		
		2.7-3.0		*Typha latifolia* (E)		
		2.7-3.0		*Typha angustifolia* (E)		
	canals	2.88, 3.64 and 4.8	N/A	*Typha angustifolia* (E)	20, 50, 65% killed	Stankovic *et al.* 1982
				Typha latifolia (E)	15, 55, 65% "	
				Phragmites communis (E)	60, 75, 80% "	
				Juncus effusus (E)	25, 60, 65% "	
	ditches	3.36	N/A	*Nymphaea odorata* (Fl)	100% killed	Welker & Reimer 1973
				Nuphar sp. (Fl)	100% "	

[a] The letters following each species name refer to the vegetation type: (E) for emergent vegetation, (S) for submergent vegetation and (Fl) for floating vegetation

Table 7.10. Recommended application rates of six herbicides for control of aquatic and crop weeds.

Herbicide	Target Species	Recommended Rates of Application[a] (kg a.i. ha^{-1})	Rates for Crop Weed Control (kg a.i. ha^{-1})	Habitat
Amitrol	cattails	9.0 - 11.0	7.0	
Dalapon	cattails and bullrushes	11.1 - 18.5	1.3 - 12.6	
Diquat	submergents and duckweed	4.4 - 7.0	0.55	farm pond, lake, canal
Glyphosate	cattails	2.5 - 4.3	1.2 - 4.3	
Paraquat	cattails and bullrushes	1.1 - 2.2	1.1	
Paraquat + Diquat	submergents	1.1 - 2.2 + 1.1 - 2.2	1.1 0.55	ditches
Simazine	algae submergents *Chara*	4.5 9.0 18.0	2.1 - 6.4	farm pond
	rooted aquatic weeds	21.6	2.1 - 6.4	ditches

[a] From: "Guide to chemical weed control", Manitoba Agriculture, 1983.

controlling aquatic vegetation, one can totally eliminate the weeds without having to avoid damage to a crop. It is therefore difficult to evaluate the impact of the application of herbicides on wetland vegetation in the Canadian pothole region.

Another set of studies, which date back to the 1950s and early 1960s and deal with the toxicity of herbicides to aquatic organisms, were summarized and tabulated by Lawrence (1962). Table 7.11 lists that information which deals with the toxicity to macrophytes of three herbicides relevant to this report. However, the compounds were applied aerially at very high rates to control aquatic weeds and thus the results are of little use in evaluating the effect of herbicides on macrophytes.

Table 7.12 summarizes the results of a number of laboratory and field studies on the effects of seven herbicides on aquatic vascular plants. The former focus on the inhibitory effects of the acute exposure of herbicides to growth or photosynthesis with the exception of two studies which look at chronic exposure. The field work was done in a variety of habitats such as ponds, enclosures, canals or lakes. The advantage of these reports over the studies presented in Table 7.9 lies in the control of actual herbicide exposure during the experiments as concentrations of the compounds in the water were measured.

The ranking of the plant species with regards to their sensitivity to herbicides is not possible due to the paucity of comparative work, incomplete data sets and the high variability of the responses. For example, the concentration of atrazine which reduced growth by 50% varied between 0.08 mg L^{-1} for Elodea and 1.0 mg L^{-1} for Myriophyllum (Forney & Davis 1981) while the concentration which inhibited photosynthesis by 50% in four species of plants only ranged between 0.08 and 0.1 mg L^{-1} (Jones & Winchell 1984). The relative sensitivity of a species also changed with the herbicide investigated: growth of Elodea was ten times more sensitive to atrazine than Myriophyllum which, in turn, was inhibited by concentrations of metribuzin half those required for Elodea (Forney & Davis 1981).

Some emergent vegetation was affected by concentrations of herbicides used for submergent control. The arrowhead, Sagittaria montividensis, was completely eliminated in ponds treated with 0.5 and 1.0 mg L^{-1} of dimethylamine 2,4-D (Boyle 1980) and cattails, Typha sp., were killed in a pond treated with 0.8 to 1.0 mg L^{-1} of diquat (Newbold et al. 1974). This suggests that emergents are susceptible to herbicides, not only when the compounds are applied to the aerial parts of the plants, but also when they contaminate the water.

Only a fraction of the major dominant wetland plant species found in the prairie pothole region are reported in the studies cited in Table 7.12. It is therefore not possible to say which species will be affected the most by herbicides through drift or contamination of slough waters.

A ranking of the herbicides reviewed in Table 7.12 on the basis of their phytotoxicity is restricted due to the lack of information. The laboratory studies indicate that 2,4-D amine, atrazine and metribuzin inhibited photosynthesis or growth at concentrations of 0.1 mg L^{-1} or less.

Table 7.11. Effects of three herbicides on emergent vegetation[a].

Herbicide	Remarks	Application Rates (kg a.i. ha^{-1})	Genus	Control or response
2,4-D acid	irrigation channel	18.4	Carex	85%
		38.1		95%
		1.12-2.24	Salix	suppressed growth
2,4-D amine	pond	3.36	Eleocharis	50%
	greenhouse	40.0	Carex	90%
	sprayed	2.24	Salix	60-80%
	sprayed	4.48		90%
		4.5	Typha	30-40%
2,4-D ester	greenhouse	4.5	Carex	10%
		5.9		10%
		48.6		75%
	pond	5.4	Eleocharis	95%
		5.8		90-100%
		4.5		90-100%
		9.0		90-100%
	pond	5.4	Juncus	90-100%
		4.5-9.0		
	pond	5.4	Polygonum	90-100%
		4.5-9.0		90-100%
	?	4.7	Pontederia	85%
	sprayed	3.4	Salix	95%
		2.24	Scirpus	95%
	pond	5.4		90-100%
		4.5-9.0		90-100%
		5.6	Typha	75%
		4.7		50-85%
		2.24		80%
	greenhouse	3.0-121.0		85-100%
	irrigation channel	6.7-9.0		100%
		3.7		99%
		5.6		90%
		16.8		80%
		4.5		30%
		17.5		80-100%
		11.2		98%
	greenhouse	1.6-121.0		75-100%
	irrigation channel	10.1-13.7		100%
	pond	5.4		90-100%

Table 7.11 (cont'd)

Herbicide	Remarks	Application Rates (kg a.i. ha^{-1})	Genus	Control or response
Amitrole	pond	7.6	*Eleocharis*	90-95%
		22.4	*Phragmites*	100%
		11.2	*Polygonum*	100%
		11.2-22.4	*Scirpus*	100%
		2.24-11.2	*Typha*	80-100%
		17.9-26.9		40%
		5.6-16.8		75-95%
	pond	6.7		100%
		13.4		39%
	pond	22.4		97%
Dalapon	pond	26.9	*Eleocharis*	100%
		17.92	*Glyceria*	100%
		22.4		75%
		14.3	*Juncus*	100%
		26.8		100%
		11.2	*Phragmites*	95%
		22.4-56.0		100%
		12.3		90%
		16.8-67.2		92-100%
		5.6-22.4	*Typha*	55-90%
		22.4		97%

[a] Extracted from Table 9 in Lawrence 1962

Table 7.12. Effects of seven herbicides on aquatic vascular plants in laboratory and field tests

Herbicide	Type of Study	Concentrations of Herbicides (mg L^{-1})		Macrophyte Species[b] and Response			Source
		used in studies	expected in ponds[a]				
Atrazine	microcosm (salinity = 5%)	0.075 and 0.65	0.4	*Potamogeton pectinatus* (S)	photos. inhibition at 0.65 mg L^{-1}		Correll and Wu 1982
				Zanichellia palustris (S)	photos. inhibition at 0.075 and 0.65		
	Laboratory growth study	0.08-1.0	--	*Elodea* sp. (S)	0.08-0.1 mg L^{-1}	EC$_{50}$	Forney and Davis 1981
				Potamogeton perfoliatus (S)	0.4-0.9		
				Myriophyllum sp. (S)	1.0		
				Vallisneria (S)	0.4-0.5		
	Laboratory study of photosynthesis	0.1	--	*Potamogeton perfoliatus* (S)	55-69% photosynthetic inhibition		Jones and Estes 1984
	Laboratory study of photosynthesis	0.077-0.104	--	*Potamogeton perfoliatus* (S)	0.077 mg L^{-1}	photosynthetic I$_{50}$	Jones and Winchell 1984
				Myriophyllum spicatum (S)	0.104		
				Zannichellia palustris (S)	0.091		
				Ruppia maritima (S)	0.102		
2,4-D	Laboratory study of continuous sustained exposure	0.03-0.25	0.09-0.26	*Myriophyllum spicatum* (S)	0.05-0.1 mg L^{-1}	sustained concentrations needed to attain 0 growth	Westerdahl and Hall 1983
				Potamogeton pectinatus (S)	0.1-0.25		
2,4-D amine	Pond study	1.0	0.1-0.26	*Myriophyllum spicatum* (S)	100% killed		Scott *et al*. 1981
				Elodea canadensis (S)			
				Potomogeton pectinatus (S)	inhibited		
				P. foliosus (S)			
	Laboratory growth study	0.045-0.27	--	*Myriophyllum spicatum* (S)	EC$_{50}$ = 0.053 mg L^{-1}		Stanley 1975

Table 7.12 (cont'd)

Herbicide	Type of Study	Concentrations of Herbicides (mg L^{-1})		Macrophyte Species[b] and Response		Source
		used in studies	expected in ponds[a]			
2,4-D ester	Pond study	1.0	0.09-0.2	*Myriophyllum spicatum* (S)	100% killed	Scott *et al.* 1981
				Elodea canadensis (S) *Potamogeton pectinatus* (S) *P. foliosus* (S)	inhibited	
2,4-D butoxyethanol ester	Laboratory chronic exposure study	0.01, 0.1 and 1.0		*Myriophyllum spicatum* (S) *Vallisneria americana* (S) *Elodea canadensis* (S)	mean time to reach 90% mortality at 0.01 mg L^{-1}, 0.1 and 1.0: 24, 19 and 15 days	Quinn *et al.* 1977
2,4-D dimethylamine	Pond study	0.5 and 1.0	0.1-0.26	*Sagittaria montividensis* (E)	100% killed	Boyle 1980
Diquat	Laboratory still water test	1.0	0.068	*Elodea densa* (S) *Najas guadalupensis* (S) *Ceratophyllum demersum* (S)	> 60% killed after 24 hours	Blackburn 1963
	Field study drainage channels	1.0		*Elodea canadensis* (S) *Myriophyllum spicatum* (S)	100% killed	Marshall 1984
				Potamogeton pectinatus (S)	slow decline in standing crop	
	Pond study	0.8-1.0		*Chara decliculata* (S) *Typha latifolia* (E) *Potamogeton gramineus* (S)	decayed after 8 days	Newbold *et al.* 1974
	Pond study	1.0		*Elodea canadensis* (S)	100% killed	Peverly and Johnson 1979
	Laboratory chronic exposure study	0.01, 0.1 and 1.0		*Myriophyllum spicatum* (S) *Vallisneria americana* (S) *Elodea canadensis* (S)	mean time to reach 90% mortality at 0.01 mg L^{-1}, 0.1 and 1.0: 16, 15 and 10 days	Quinn *et al.* 1977

Table 7.12 (cont'd)

Herbicide	Type of Study	Concentrations of Herbicides (mg L^{-1})		Macrophyte Species[b] and Response		Source
		used in studies	expected in ponds[a]			
Linuron	Enclosures in pond	1.0	0.03-0.26	*Potamogeton crispus* (S) *P. pectinatus* (S) *P. natans* (S) *Chara vulgaris* (S)	did not develop in treated enclosures	Stephenson and Kane 1984
Metribuzin	Laboratory growth study	0.08-1.0	0.032	*Myriophyllum* sp. (S) *Elodea* sp. (S)	0.04 mg L^{-1} 0.08 mg L^{-1} EC_{50}	Forney and Davis 1981
Paraquat	Canal	1.0	0.135	*Ceratophyllum demersum* (S)	50% reduction	Arsenovic *et al*. 1982
	Laboratory growth study	1.0	--	*Elodea canadensis* (S) *Ceratophyllum demersum* (S)	100% killed	Best and Wittenboer 1978
	Laboratory still water tests	1.0	--	*Elodea densa* (S) 35 *Najas guadalupensis* (S) 70 *Ceratophyllum demersum* (S) 50	% killed after 24 hours	Blackburn 1963
	Reservoir	1.0	--	*Potamogeton pectinatus* (S) *Myriophyllum spicatum* (S)	100% killed	Brooker and Edwards 1973
	Pond	0.175	--	*Elodea canadensis* (S)	100% killed	Pokorny *et al*. 1971
	Lake	0.5	--	*Elodea canadensis* (S) *Myriophyllum spicatum* (S)	100% killed	Water Pollution Research Laboratory 1967
	Lake	0.5	--	*Elodea* sp. (S) *Myriophyllum* sp. (S) *Potamogeton* sp. (S)	100% killed	Way *et al*. 1971

Table 7.12 (cont'd)

Herbicide	Type of Study	Concentrations of Herbicides (mg L^{-1})		Macrophyte Species and Response		Source
		used in studies	expected in ponds			
	Lake	0.5	--	*Elodea* sp. (S) *Myriophyllum* sp. (S) *Potamogeton pectinatus* (S) *Zanichellia* sp. (S)	100% killed	Way et al. 1971
				Typha sp. (E)	some damage	
Simazine	Lake	1.9	0.26-0.79	*Lemna minor* (Fl) *Ceratophyllum* (S) *Potamogeton* sp. (S) *Nuphar advena* (Fl) *Nymphaea tuberosa* (Fl)	high degree of control	Ahrens and Block 1972
	Pond	1.0	0.26-0.79	Almost all submerged macrophytes were eliminated		Gorden et al. 1982

[a] From Table 7.7

[b] The letters following each species name refer to the following vegetation types: (E) for emergent vegetation, (S) for submergents, and (Fl) for floating and floating-leaved plants

The field studies showed that paraquat and 2,4-D dimethylamine resulted in 100% die-off of certain macrophytes at concentrations of 0.5 and 0.175 mg L^{-1}, respectively. One cannot infer, however, that these herbicides are the most toxic to aquatic plants because no systematic comparative study has been done to determine the "no effect" concentrations of the herbicides used in the prairies. However, the concentrations at which the four compounds mentioned above were found to produce an effect are close to the maximum initial concentrations expected after direct overspray (see Table 7.7). Of potentially greater concern is the problem of chronic exposure. Quinn and co-workers (1977) demonstrated 90% mortality in three species of submergent vegetation after exposure for 24 and 16 days to 0.01 mg L^{-1} of 2,4-D BEE and diquat, respectively. While 2,4-D is not a very persistent compound and may not be found at such concentrations for that period of time, compounds which combine high phytotoxicity and moderate to long persistence, such as the triazines and the bypiridilliums (e.g. diquat, paraquat), may have significant impact on pothole vegetation when used on a yearly basis.

7.2.2 Indirect and long-term impact

Plant die-off in or around aquatic ecosystems is the first in a series of events which may follow exposure to a highly phytotoxic and moderately to highly persistent herbicide. This section will attempt to pinpoint what these events may be in the context of prairie potholes subject to herbicide drift and/or runnoff. Because there is no information on this specific question, one must extrapolate from the body of literature available on the ecological effects of chemical aquatic weed control programs. The effect of herbicides in aquatic systems has been well documented in a number of reviews (Brooker & Edwards 1975, Hurlbert 1975, Newbold 1975, Wade 1981). Therefore, the approach followed herein will begin with a review of the response to aquatic ecosystems to herbicide inputs which focuses specifically on the interplay between the biological and physico-chemical events following herbicide exposure, the time scales involved and the recovery process. From this, one can isolate the key factors determining the response of the community and the direction of change. An understanding of these factors will then permit the development of various scenarios of events which may follow the contamination of prairie potholes by herbicides.

7.2.2.1 Ecosystem response

The immediate effect of most herbicides applied to control aquatic weeds is to inhibit photosynthesis. This often leads to decreased dissolved oxygen (D.O.) concentrations as early as two days after application (Pokorny et al. 1971, Newbold et al. 1974) and lasting as long as 46 days (Gorden et al. 1982). While the initial drop in D.O. is the result of photosynthetic inhibition, low levels are maintained by the increased respiratory activity of the bacterial populations involved in the breakdown of the dead plant material. For example, D.O. levels in a pond treated with diquat decreased two days after treatment and the biological oxygen demand did not increase until eight days after treatent (Newbold et al. 1974). In a reservoir treated with paraquat to control weeds, Brooker and Edwards (1973) found treatment, the range of diel D.O. values went from 70-150% to 70-100% of saturation. Decreases in oxygen concentrations after herbicide treatment may not always be the result of herbicide inhibition of photosynthesis or bacterial activity: high densities of a replacement species, the duckweed Lemna minor, were related to low D.O. levels due to shading which retarded the growth of phytoplankton and submergent vegetation (Brooker 1976). It is not uncommon to fail to record changes in oxygen concentration following herbicide application. A lack of response found by Scorgie (1980) was attributed to the slow release rate of the herbicide cyanatrin applied as a granule, and to the apparent inhibition of bacterial decomposition by this herbicide. Peverly and Johnson (1979) did not find a drop in oxygen concentrations following the application of diquat in ponds to control the weed Elodea. This was attributed to the low temperatures of the water as the treatment occurred in the fall. The partial treatment of a large reservoir to control the water milfoil, Myriophyllum spicatum, had no apparent effect on D.O. levels, possibly due to the importance of diffusion through mixing and currents in such a large body of water (Morris & Jarman 1981). While certain plant kills following herbicide treatment may lead to severe drops in D.O., these rarely last more than one or two weeks (Table 7.13) and never resulted in anoxic conditions, except possibly near the bottom of shallow ponds (Pokorny et al. 1971).

The inhibition of photosynthesis and increase in respiration lead to increased levels of CO_2 and the destruction of macrophytes may often lead to changes in a series of interrelated variables involved in the inorganic carbon balance: increases in free CO_2, bicarbonate and alkalinity along with decreases in pH and concentrations of the carbonate ion (see Hurlbert 1975). These changes have been recorded when herbicides were used in a variety of systems such as lakes (Water Pollution Research Laboratory 1967), reservoirs (Wojtalik et al. 1971, Brooker & Edwards 1973), ponds (Simpson & Pimentel 1972) or drainage channels (Marshall 1984). However, Hurlbert (1975) noted that these changes did not occur if macrophyte kills were followed by phytoplankton blooms as occured in the treatment of ponds with dichlobenil (Walsh et al. 1971). One exception to this observation is the response of fish ponds treated with paraquat to control Elodea (Pokorny et al. 1971). In this study, marked increases in alkalinity and free CO_2, along with a decrease in pH from 10.0 to 8.3 were measured at the same time a large increase in phytoplankton was noted. In brief, changes in pH, alkalinity and inorganic carbon (CO_2, $CO_3^=$, HCO_3^-) are closely tied to the inhibition of photosynthesis and increase in community respiration. All these events occur within the same time scale: usually one to two weeks following treatments (Table 7.13).

Massive plant die-offs and the subsequent bacterial breakdown can lead to substantial inputs of nutrients to the water. The loss of oxygen under such conditions can lead to anoxic conditions near the bottom which favor rapid remineralization of nitrogen and phosphorus. Increased levels of phosphorus and/or nitrogen following herbicide treatment have been observed in lake (Water Pollution Research Laboratory 1967) and pond (Gordon et al. 1982) ecosystems. These events usually occur within a month of treatment and are frequently followed by phytoplankton blooms induced by the increased input of nutrients. This chain of events is highlighted in the comparison of the response to diquat of previously fertilized ponds to

Table 7.13. Time scale of some events following application of herbicides for aquatic weed control

Herbicide (mg L^{-1})	Time (days after treatment) to				Source
	Decrease in dissolved oxygen	Changes in CO_2, $CO_3^=$, HCO_3^-, pH or alkalinity	Die-off of macrophytes	Algal bloom	
Cyanatrin (0.2)			84		Scorgie 1980
2,4-D BEE			15-21	28	Getsinger et al. 1982
2,4-D BEE (1.0)			14		Scott et al. 1981
2,4-D DMA (1.0)					
2,4-D DMA		1-14	21-28		Wojtalik et al. 1971
Dichlobenil (1.0)				32-64	Newbold 1974
(1.0)				14-21	Walsh et al. 1971
Diquat (1.0)		10-15			Marshall 1984
(1.0)	2-8		8	8-64	Newbold et al. 1974
Endothall (0.3)	3-46				Gorden et al. 1982
(5.0)	3-18				Serns 1975
Paraquat (1.0)		3	26		Brooker and Edwards 1973
(0.175)	2	5-6	15	6	Pokorny et al. 1971
(0.5)	5	5	21	21-28	Water Pollution Research Laboratory 1967
(0.5)			3		
(0.5)			32		Way et al. 1971
Simazine (1.0)	1-10	1-42			Gorden et al. 1982
(0.1)	3	7			Pruss and Higgins 1975

unfertilized ponds (Peverly & Johnson 1979). The authors found that the algae bloomed, following tissue mineralization, in proportion to the amount of phosphorus in plant tissues which, in turn, reflected levels of fertilization. The actual recording of increases in concentrations of N and P in waters treated with herbicides is rare. While inputs may increase, rapid losses through uptake by algae or sedimentation keep concentrations unchanged in the water. The rapid growth of the macrophytic alga Chara (Brooker & Edwards 1973, Scott et al. 1981) or of phytoplankton (Walsh et al. 1971) following herbicide treatment may dampen changes in nutrient concentrations. Calcium and magnesium were found to decrease in a reservoir treated with paraquat (Brooker & Edwards 1973). These cations were probably precipitated as carbonates on the surface of the alga Chara which grew rapidly after treatment. Long-term effects of macrophyte removal on nutrients has been noted in ponds treated with fenac or dichlobenil: one year after treatment, the ponds had lower standing crops of macrophytes, decreased concentrations of certain forms of N and P, and lower gross primary productivity (Boyle 1979). This is additional evidence that macrophytes act like nutrient pumps, extracting nutrients from the sediments and releasing them in the water (see Chapter 2).

The response of aquatic plants to herbicide application can be divided into three periods. First, the period of plant die-off during which the live standing crop of sensitive species drops rapidly. The second period consists of the immediate response of resistant species or replacement vegetation favored by the change in conditions. This intermediate stage may have a bearing on the third part of the response: the recovery of the system to its previous state.

The time interval between the application of the herbicide and the beginning of plant breakdown, after it has collapsed to the bottom of the water body, is usually about one to two weeks and is rarely longer than one month (Table 7.13). The response of the remaining vegetation to this disturbance will depend on the extent of the die-off and the changes in physical and chemical conditions, and on the presence or absence of resistent species. Few species are usually left unscathed after the use of herbicides to control aquatic weeds, however, three species seem to be relatively tolerant. The submergent Potamogeton crispus survived the use of 2,4-D in ponds (Scott et al. 1981) and has been found to replace, within a few months, species which had been eliminated by cyanatryn (Scorgie 1980) and paraquat (Way et al. 1971). The emergent Polygonum amphibium was not affected by the treatment of ponds with diquat (Newbold 1974) nor was it killed by the application of paraquat to a lake (Way et al. 1971). The third species, the macrophytic alga Chara was only found to be resistant to endothall (Serns 1975) but it often dominates after the use of herbicides for aquatic weed control.

The algae, either phytoplankton, benthic filamentous algae or the large stoneworts (e.g. Chara or Nitella) are usually the first to benefit from the changed conditions. This usually occurs within one to two months (Table 7.13). The sudden bloom of algae is most probably the result of changes in nutrient availability, the amount of light entering the water column and free space on the substrate. The high intrinsic growth rate of algae compared to that of vascular plants allows them to recover much more quickly from herbicide toxicity. Phytoplankton blooms occurred six days after the treatment of ponds with paraquat (Pokorny *et al*. 1971), 32 days after ponds were treated with dichlobenil (Newbold 1974) and in the fourth week after 2,4-D BEE was added to a shallow bay (Getsinger *et al*. 1982). Concentrations of chlorophyll a peaked 21 to 28 days after the water of a lake treated with paraquat (Water Pollution Research Laboratory 1967). Benthic filamentous algae have been known to bloom after the elimination of macrophytes (Crawford 1981, Newbold *et al*. 1974) and may possibly play a role in making conditions favorable for the establishment of *Chara*. It has been suggested that benthic algae stabilize the bottom sediments and thus, by reducing turbidity, allow the development of *Chara* (Crawford 1981). The numerous reports of the dominance of stoneworts after the elimination of vascular plants by herbicides (Brooker & Edwards 1973, Newbold *et al*. 1974, Crawford 1981, Scott *et al*. 1981, Getsinger *et al*. 1982) are an indication not only of their ability to colonize substrates in the early stages of succession, but also of their importance in setting the stage for the regrowth of vascular plants. It has been suggested that the growth of *Nitella* or *Chara* decreases the turbidity of the water and therefore favors the return of other macrophytes (Crawford 1981, Getsinger *et al*. 1982). This may be accomplished through the stabilization of the sediments and the inhibition of phytoplankton *via* competition for nutrients.

This stage of dominance by algae or opportunistic species lasts until conditions are appropriate for the recolonization by aquatic macrophytes. This may take a few months (Crawford 1981) to a few years (Getsinger *et al*. 1982). Newbold (1975) tabulated the time required for the regrowth of 43 aquatic plant species killed by eight herbicides (Table 7.14). Information existed on 169 of the 344 possible plant-herbicide combinations. Examination of these responses for all plant species shows that in 96.4% of the cases the plant regrew within one year or less. Wade (1982), in reviewing long-term effects of herbicides on aquatic weed communities, concluded that revegetation after herbicide treatment will usually lead to communities very similar in species composition and abundance to the one present before treatment. When changes in species composition did occur, they usually involved rare species replacing other uncommon species. He hypothesized that root systems and propagules left in the sediments were unscathed by the herbicides. This may explain the lack of long-term changes in the aquatic plant communities. Crawford (1981) found that seeds and tubers of *Potamogeton foliosus* in a pond survived treatment with simazine and thus allowed rapid regrowth of the plants less than one month after treatment.

7.2.2.2 Factors determining community response

In order to evaluate the potential damage of herbicide contamination on prairie potholes, one must isolate the variables which determine the extent of the impact. A review of the effects of herbicides on aquatic plants points to five important factors determining the response of the community: the size of the body of water, the type of herbicide used, the route of entry, the time of year and the extent of plant mortality.

Table 7.14. Recorded effects of herbicides on aquatic plants^{ab}.

Plants Submergent and Floating Species		Herbicides for Submergent and Floating Species					Herbicides for Emergent Species		
Botanical name	Common English name	Diquat	Paraquat	Dichlobenil	Chlothiamid	Terbutryne	Dalapon	Dalapon Paraquat	2,4-D
Ranunculus sp.	Water crowfoot	R	R	K1	K1	K1	N	N	N
Nymphaea alba (L.)	White water lily	R	R	R	R	N	N	N	R
Nuphar lutea (L.)	Yellow water lily	R	R	R	R	N	N	N	R
Rorippa nasturtium-aquaticum (L.)	Watercress	K1	K1	K1	K1	K1	N	N	K1
Myriophyllum spp.	Water milfoil	K1	K1	K1	K1	K1	N	N	N
Callictriche spp.	Starwort	K1	K1	K1	K1	K1	N	N	N
Ceratophyllum spp. (L.)	Horn-wort	R	R	K1	K1	K1	N	N	N
Hottonia palustris (L.)	Water violet	K1	K1	K1	K1	K1	N	N	N
Utricularia tulgaris (L.)	Greater bladderwort	O	O	R	R	O	N	N	N
Polygonum amphibium (L.)	Amphibious bistort or persicaria	T	T	R	R	N	N	N	R (high doses only)
Sagittaria sagittifolia (L.)	Arrowhead	K1	K1	K1	K1	K1	N	N	N
Hydrocharis morsusranae (L.)	Frog-bit	K1	K1	K1	K1	K1	N	N	N
Elodea canadensis (Michx)	Canadian pondweed	K1	K1	K1	K1	K1	N	N	N
Lemna trisulca (L.)	Ivy duckweed	R	R	K1	K1	K1	N	N	N
Lemna minor (L.)	Lesser duckweed	T	N	N	N	K1	N	N	N
Potamogeton natans (L.)	Floating pondweed	K1	K1	K1	K1	R	N	N	N
Potamogeton lucens (L.)	Shining pondweed	K1	K1	K1	K1	K1	N	N	N
Potamogeton gramineus (L.)	Various-leaved pondweed	K1	K1	K1	K1	K1	N	N	N
Potamogeton crispus (L.)	Curled pondweed	K1	K1	K1	K1	K1	N	N	N
Potamogeton pectinatus (L.)	Fennel pondweed	K1	K1	K1	K1	K1	N	N	N
Glyceria fluitans (L.)	Floating sweet grass	N	N	K1	K1	K1	R	N	N
Chara spp.	Stoneworts	R	R	K2	K2	K1	N	N	N
Algae									
Vaucheria spp.	Blanket weed Cott	N	N	N	N	K1	N	N	N
Cladophora spp.	Blanket weed	K1	N	N	N	K1	N	N	N
Spirogyra spp.	Blanket weed	R	N	N	N	K1	N	N	N
Enteromorpha intestinalis (L.)		N	N	N	N	K1	N	N	N
Diatoms		N	N	N	N	N	N	N	N

Table 7.14 (cont'd)

Plants Submergent and Floating Species		Herbicides for Submergent and Floating Species					Herbicides for Emergent Species		
Botanical name	Common English name	Diquat	Paraquat	Dichlo-benil	Chlo-thiamid	Terbutryne	Dalapon	Dalapon Paraquat	2,4-D
Emergent species									
Hippuris tulvaris (L.)	Mare's-tail	R	R	O	O	K1	N	N	N
Lythrum salicaria (L.)	Purple loosestrife	N	N	N	N	N	N	N	R
Oenanthe spp.	Water dropwort	O	O	K1	K1	O	N	N	R
Mentha aquatica (L.)	Water mint	N	N	N	N	N	N	N	K1
Rumex hydrolapathum (Hudson)	Greater water dock	N	N	N	N	N	N	N	T
Alisma plantago-aquatica (L.)	Water plantain	K1	K1	K1	K1	K1	N	R	K1
Iris pseudacorus (L.)	Yellow flag	R	R	K1	K1	R	N	N	N
Juncus eflueus (L.)	Soft rush	T	T	N	N	N	R	K1	T
Typha latifolia (L.)	Reedmace	R	K1	K1-	K1-	N	K2	K1	N
Sparganium erectum (L.)	Branched bur reed	K1	K1	R	R	N	K1	K2	K1
Schoenoplectus lacustris (L.)	Bulrush	N	N	N	N	N	T	K1	N
Carex riparia (Curtis)	Great pond sedge	N	N	N	N	R	R	K1	N
Phragmites communis (Trin.)	Reed	N	N	N	N	N	K3+	K3+	N
Agrostis stolonitera (L.)	Creeping bent	R	R	K1	K1	O	O	O	O
Phalaris arundinacea (L.)	Reed canary grass	N	N	N	N	N	K1	K1	N
Glyceria maxima (Hartman)	Reed sweet grass	N	N	N	N	N	R	N	N
Equisetum flutiatile (L.)	Water horsetail	R	R	K1	K1	R	N	N	R

[a] Key:

K1	= Kill for one year
K2, etc.	= Kill for two years, etc.
R	= Regrowth in the same season
T	= Temporary kill for 2 to 4 weeks
N	= Not recommended for use, likely to be inefficient
O	= No information available

[b] Taken from Newbold 1975

The size of the water body will influence the extent of the contamination of a pothole. Drift onto a large pothole will be of lesser significance than drift onto a smaller slough because dilution through mixing and diffusion will be greater in the former system. Also, in larger sloughs, the chances of recolonization from within the system will be more likely.

The type of herbicide used will determine the extent of the dieoff, the species affected and the persistence of the toxic effects.

The route of entry of the herbicide should have some bearing on the types of changes expected. If a pothole is oversprayed, or if the herbicide drift onto the slough is severe, damage to the emergent component of the vegetation will be expected. However, if runoff is the major mode of entry, contamination may not be as acute but will probably affect both submergents and emergents.

The time of year that the herbicide is applied will have an effect on its persistence, efficacy and the subsequent impact because the temperature of the water and its oxygen content will influence the chemical behaviour of the herbicide. These two parameters will also determine the rates of bacterial breakdown of the decaying vegetation and subsequently the water chemistry of the pothole. In addition, aquatic plants are more susceptible to herbicides in the early stages of growth in spring.

The extent of the mortality resulting from contamination by herbicides will, in turn, affect the physical and chemical characteristics of the slough along with the recovery. Any vegetation remaining alive following herbicide impact should buffer the changes in the slough and allow faster recovery.

7.2.2.3 Scenarios for prairie potholes

Patterson (1973) pointed out that damage by herbicide drift to sensitive crop species and shelterbelts has been documented in agricultural areas and that similar drift onto wetlands would occur and possibly cause some damage. As there is practically no experimental work on the effects of herbicide drift on potholes, it is necessary to develop the most probable scenarios of the events following contamination, based on the preceding review and knowledge of the use patterns of herbicides in the prairie pothole region of Canada.

The direct overspray of a pothole during aerial application of a herbicide may be the most damaging form of contamination with respect to acute exposure. In small shallow potholes where the vegetation is dominated by dense stands of emergents, direct overspray could lead to severe damage because the whole slough would be affected and the aerial parts of the plants would receive the bulk of the application. The severity of the impact would depend on the toxicity of the herbicide used at rates recommended for crop weed control. If overspray occurred over a larger slough, the impact could be diminished by dilution of the herbicide were it to reach areas of open water. Also, the entire slough may not be oversprayed.

The second possible scenario is that of lateral drift of this happening is high in agricultural areas as herbicides are applied yearly throughout the prairies. The consequences of such drift are difficult to evaluate as they are a function of pothole size, the distance between the pothole and the applicator and the variables which affect drift such as wind speed and temperature (see Chapter 5). Once again, the emergent vegetation may be more severely affected than the submergents due to the direct deposition of the herbicides onto the aerial parts of the plants. The effect on the submergents will depend on the amount of contaminant reaching the water and the dilution factor.

While there is little information available to develop the third scenario, it may be the most significant with respect to long-term impact. The importance of chronic exposure to low concentrations of herbicides by pothole vegetation is largely unknown. It is generally thought that inputs through runoff are usually small, although they may be substantial on occasion (see Chapter 5). Runoff input is especially critical for compounds which are persistent such as diquat, paraquat or linuron. They may be released from the sediments at a later time or maintain constant low concentrations. Goldsborough and Robinson (1983) showed that simazine and terbutryne desorbed from sediments into contaminent-free waters of previously treated enclosures in a marsh; this, in turn, slowed or even prevented the recovery of the periphyton in the enclosures. This kind of low concentration chronic exposure may affect both submergents and emergents equally. If one species was more susceptible than the others to chronic exposure to herbicides, changes in species composition would be expected. Brooker (1976) showed that when the dominant emergent Phragmites communis was maintained at very low levels in drainage channels through chemical control, the previously less common Lemna minor, Callitriche stagnalis and Potamogeton crispus became very abundant. While one would expect low concentrations of herbicides to reduce the productivity of aquatic plants, this may not be the case. The acute exposure of a pond ecosystem to 0.5 mg L^{-1} of endothall resulted in a change in the species which dominated six weeks after treatment but not in the standing crop of dominant plants (Walker 1971). Althouth this study did not examine chronic exposure, it does indicate that such an exposure may only result in the replacement of sensitive species by more resistant ones. The lack of substantial long-term impact of herbicides on plants is supported by two studies which focused on the long-term effects of herbicide use in the vicinity of drainage ditches (Heckman 1981) and in the control of vegetation in drainage channels (Wade & Edwards 1980). Both studies determined that the very minor changes which did take place in the plant communities were not the result of exposure to herbicides, but rather of other cultural impacts such as grazing by cattle, increases in salinity due to the regulation of tidal waters or loss of habitat (Wade & Edwards 1980).

7.2.3 Problems with data sets

The development of the three possible scenarios of prairie pothole contamination by herbicides points to weaknesses in the data base and obvious research needs:

(1) There is a need to do at least a minimum amount of comparative toxicity testing of the main use herbicides on the most important wetland species. This work should be done in a way such that results can be compared between studies. While efficacy data is available on some plant species targeted for control, these represent only a fraction of the macrophytes common in prairie potholes. In addition, the available studies look at the toxicity of herbicides at rates recommended for aquatic weed control, which are higher than the amounts which may actually be reaching the potholes.

(2) Studies should be aimed at monitoring the behaviour of herbicide drift from the moment it is applied in areas of high pothole densities. Thus, if herbicides are affecting wetland vegetation, the frequency of such events and the extent of damage and recovery could be related to the characteristics of herbicide use.

(3) Because potholes are self-contained bodies of water acting as "sinks" in the prairie landscape, there is a very high probability that they will accumulate residues of the more persistent herbicides. It is, therefore, crucial to evaluate the inputs and the subsequent behaviour of the herbicide. The persistence, sorption properties and degradation of herbicides should be determined under the low temperatures and frequent anoxic conditions characteristic of prairie potholes. If these compounds are found to be present and active in sloughs, the effects of chronic exposure to such levels of contamination on wetland vegetation should be determined.

7.3 Herbicide damage to waterfowl food resources

Food resources for waterfowl include both plant and animal species inhabiting prairie sloughs as well as terrestrial plants and invertebrates. The importance of various taxa in the diet of duck species was summarized in Section 2.5. In this section, those species selected as prey by ducks (both plant and animal) are examined in terms of their direct and indirect responses to herbicide toxicity. In Chapter 6 of this report, a strong case was presented for interferences with duckling growth and survival under conditions of a severely insecticide-impacted aquatic invertebrate population. Here, we analyze the parallel case of the impact of toxic herbicides on invertebrate prey for ducks, as well as the potential for herbicide elimination of plant food resources.

7.3.1 Herbicide impact on invertebrate prey

7.3.1.1 Laboratory toxicity data

For those herbicides for which laboratory toxicity data is available (19 of 32 compounds for aquatic invertebrates and 25 of 32 for fish), there are few indications of potential impacts to aquatic fauna from single overspray events. Toxicity data sets for zooplankton, amphipods and isopods, insects and aquatic snails and worms are summarized in Tables 7.15, 7.16, 7.17 and 7.18, respectively. Herbicides, in general, are not as toxic

Table 7.15. Summary of selected toxicity data for herbicide effects on zooplankton.

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]		Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
amitrole (T)	waterflea, Daphnia magna	1st instar	S	21°C	48 h TL_{50}	30	Sanders 1970
(50% AI)	"	--	S	--	48 h IC_{50}	3.55	Bunting & Robertson 1975
	"	1st instar	S	21°C	26 h EC_{50} immobilization	23 (15.3-44.0)	Crosby & Tucker 1966
(T)	"	--	S	21°C	26 h EC_{50} immobilization	40 (14.3-112)	"
	copepod, Cyclops vernalis	0-4 h nauplii	S	20°C	48 h EC_{50} immobilization	58.5 (39.8-124.0)	Robertson & Bunting 1976
	"	0-4 h nauplii	S	20°C	96 h EC_{50} immobilization	22.1 (18.5-27.7)	"
(T)	"	0-4 h nauplii	S	20°C	48 h EC_{50} immobilization	2.3 (1.8-3.9)	"
	"	0-4 h nauplii	S	20°C	96 h EC_{50} immobilization	2.0 (1.9-2.4)	"
atrazine	waterflea, Daphnia magna	< 24 h	S	20°C	48 h LC_{50}	6.9 (5.2-8.1)	Macek et al. 1976
	"	--	S	20°C	21 d reproduction	1.15	"
	"	--	S	--	48 h LC_{50}	3.6	USDI 1968
	Daphnia pulex	< 24 h	S	23°C	18 h LC_{50} (0.4% methanol stress)	0.6 (0.2-14.4)	Bowman et al. 1981
	"	--	S	--	28 d reproduction	1.0	Schober & Lampert 1977
2,4-D (acid)	waterflea, Daphnia magna	1st instar?	S	21°C	48 h TL_{50}	>100	Sanders 1970
(dimethylamine salt)	"	1st instar	S	21°C	48 h TL_{50}	4.0	"
(PGBE)	"	1st instar	S	21°C	48 h TL_{50}	0.1	"
(BEE)	"	1st instar	S	21°C	48 h TL_{50}	5.6	"
(acid)	"	1st instar	S	21°C	26 h EC_{50} immobilization	>100	Crosby & Tucker 1966
(acid)	"	3.2-3.4mm/adult	S	20°C	96 h LC_{50}	417	Presing 1981
(Na salt + dichlorophenyl)	"	3.2-3.4mm/adult	S	15°C	24 h LC_{50}	507	"
	"	3.7-3.4mm/adult	S	15°C	48 h LC_{50}	372	"
(Na salt)	"	3.2-3.4mm/adult	S	15°C	96 h LC_{50}	268	"
(Na salt)	"	3.2-3.4mm/adult	S	20°C	96 h LC_{50}	1042	"
(PGBE)	Daphnia pulex	--	S	17.5°C	3 h LC_{50}	>40	Hashimoto & Nishiuchi 1981
(acid?)	"	--	S	--	48 h LC_{50}	3.6	FWPCA 1968
(acid?)	Daphnia lumholtzi	--	S	--	38 h LC_{50}	10	George et al. 1982
(Na salt)	Moina macrocopa	--	S	17.5°C	3 h LC50	>40	Hashimoto & Nishiuchi 1981

Table 7.15 (cont'd)

Herbicide[a]	Species Tested	Size/Stage	Test Conditions	Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
2,4-D (acid)	copepod, *Cyclops vernalis*	0-4 h nauplii	S 20°C	48 h EC_{50} immobilization	37.4	Robertson & Bunting 1976
	"	0-4 h nauplii	S 20°C	96 h EC_{50} immobilization	8.7 (5.3-11.6)	"
	"	0-4 h nauplii	S 20°C	48 h EC_{50} immobilization	662.1	"
(alkanolamine)	"	0-4 h nauplii	S 20°C	96 h EC_{50} immobilization	142.0 (120.3-167.3)	"
and	*Mesocyclops leuckarti*	--	S --[†]	30 d no effect conc.	1.0	George *et al.* 1982
(BEE)	harpacticoid, *Nitocra spinipes*	adult	S 21°C	96 h LC_{50}	3.1 (2.4-4.1)	Linden *et al.* 1979
(acid?)	rotifer, *Brachionus calyciflorus*	--	S --	24 h LC_{50}	5	George *et al.* 1982
dalapon	waterflea, *Daphnia magna*	2-26 h	S 19°C	26 h LC_{50}	6	Frear & Boyd 1967
	"	--	S --	48 h LC_{50}	6	USDI 1968
(Na salt)	*Daphnia pulex*	1st instar	S 15.5°C	48 h EC_{50} immobilization	11 (8.2-14.7)	Sanders & Cope 1966
	Simocephalus serrulatus	1st instar	S 15.5°C	48 h EC_{50} immobilization	16 (11.4-22.4)	"
(Na salt)	copepod, *Cyclops* sp.	--	S 27°C	24 h no mortality	200 (a.e.)	cf. Kenaga 1974
	"	--	S 27°C	24 h 100% mortality	400 (a.e.)	"
	Heliodiaptomus vidus	--	S --	323 h LC_{50}	1	George *et al.* 1982
	harpacticoid, *Nitocra spinipes*	adult	S 21°C	96 h LC_{50}	4800 (3970-5810)	Linden *et al.* 1979
dicamba	waterflea, *Daphnia magna*	1st instar	S 21°C	48 h TL_{50}	>100	Sanders 1970
(Banvel®)	"	--	S --	48 h LC_{50}	111	cf. Ghassemi *et al.* 1981
	Daphnia pulex	1st instar	S 15.5°C	48 h EC_{50} immobilization	11	Sanders & Cope 1966
diclofop-methyl	waterflea, *Daphnia magna*	--	S --	48 h LC_{50}	4.03	Weed Society of America 1983
diquat	waterflea, *Daphnia magna*	1st instar	S 21°C	26 h EC_{50} immobilization	7.1 (6.3-8.0)	Crosby & Tucker 1966
	Daphnia pulex	--	S --	8 d LC_{50}	1.0	Gilderhus 1967

Table 7.15 (cont'd)

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]		Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
flamprop-methyl	waterflea, *Daphnia magna*	--	S	--	24 h LC_{50}	15.1 (14.0-16.4)	Weed Society of America 1983
glyposate (tech)	waterflea, *Daphnia magna*	--	S	--	48 h LC_{50}	780	Weed Society of America 1983
(Roundup)	"	--	S	--	48 h LC_{50}	5.3	"
(Roundup)	"	--	S	22°C	48 h EC_{50} immobilization	3.0	Folmar *et al*. 1979
	harpacticoid, *Nitocra spinipes*	adult	S	21°C	96 h LC_{50}	22 (17-29)	Linden *et al*. 1979
linuron	waterflea, *Daphnia magna*	--	S	--	24 h LC_{50}	4	Knauf & Schulze 1972
MCPA (acid)	waterflea, *Daphnia magna*	1st instar	S	21°C	26 h LC_{50} immobilization	>100	Crosby & Tucker 1966
paraquat	waterflea, *Daphnia magna*	1st instar	S	21°C	26 h EC_{50} immobilization	11.0 (9.1-12.2)	Crosby & Tucker 1966
	Daphnia pulex	--	S	--	3 h TLm	>40	Hashimoto & Nishiuchi 1981
	"	1st instar	S	15.5°C	48 h EC_{50} immobilization	3.7 (2.8-4.8)	Sanders & Cope 1968
	Simocephalus serrulatus	1st instar	S	15.5°C	48 h EC_{50} immobilization	4 (2.7-6.0)	"
	Moina macrocopa	--	S	--	3 h TLm	>40	Hashimoto & Nishiuchi 1981
picloram (acid)	waterflea, *Daphnia magna*	1st instar	S	20°C	24 h LC_{50}	50.7 (44.7-57.7)	Mayes & Dill 1984
(K salt)	"	1st instar	S	20°C	48 h LC_{50}	212 (180-253)	"
(acid)	"	1st instar	S	20°C	48 h LC_{50}	63.8 (63-75)	Gersich *et al*. 1985
(TIPA salt)	*Daphnia* sp.	--	S	--	24 h no effect conc.	380	Lynn 1965
	"	--	S	--	10 w no effect conc.	1	Hardy 1966
propanil	waterflea, *Daphnia magna*	1st instar	S	21°C	26 h EC_{50} immobilization	4.8 (3.8-6.6)	Crosby & Tucker 1966
	Daphnia pulex	--	S	--	3 h TLm	>40	Hashimoto & Nishiuchi 1981
	Moina macrocopa	--	S	--	3 h TLm	>40	"

Table 7.15 (cont'd)

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
simazine	waterflea, Daphnia magna	1st instar	S 21°C	48 h TL_{50}	1.0	Sanders 1970
	Daphnia pulex	--	S --	3 h TLm	>40	Hashimoto & Nishiuchi 1981
(aquazine®)	"	24 h	S 20°C	48 h LC_{50}	98.1	Fitzmayer et al. 1982
	Moina macrocopa	--	S --	3 h TLm	>40	Hashimoto & Nishiuchi 1981
TCA	waterflea, Daphnia magna	--	S --	48 h LC_{50}	2000	Dennis et al. 1979
triallate	waterflea, Daphnia magna	--	S --	48 h LC_{50}	0.43	Weed Society of America 1983
trifluralin	waterflea, Daphnia magna	1st instar	S 21°C	48 h TL_{50}	0.56	Sanders 1970
	Laphnia pulex	1st instar	S 15.5°C	48 h EC_{50} immobilization	0.23 (0.16-0.36)	Sanders & Cope 1966
	"	--	S --	2 h TLm	>40	Hashimoto & Nishiuchi 1981
	Moina macrocopa	--	S --	3 h TLm	>40	"
	Simocephalus serrulatus	1st instar	S 15.5°C	48 h EC_{50} immobilization	0.45 (0.33-0.62)	Sanders & Cope 1966
(Treflan®)	Alonella sp.	--	S 20°C	48 h LC_{50}	0.06	Naqvi et al. 1985
	copepod, Diaptomus sp.	--	S 20°C	48 h LC_{50}	0.08	"
	Eucyclops sp.	--	S 20°C	48 h LC_{50}	0.05	"

[a] Formulation indicated in brackets

[b] S = static

Table 7.16. Summary of toxicity data for herbicide effects on aquatic amphipods, isopods and ostracods.

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
amitrole	*Gammarus fasciatus*	~ 2 months	S 15.5°C	48 h TL_{50}	> 100	Sanders 1970
	Asellus brevicaudus	adult	S 15.5°C	48 h TL_{50}	> 100	"
	Cypridopsis vidua	juvenile	S 21°C	48 h TL_{50}	32	"
atrazine	*Gammarus fasciatus*	adult	S 20°C	48 h LC_{50}	5.7 (3.6-8.0)	Macek *et al*. 1976
	"	--	S 20°C	30 d reproductive	0.14	"
	Hyalella azteca	juvenile	S 23°C	18 h LC_{50}	8.8 (7.2-18.6)	Bowman *et al*. 1981
2,4-D (dimethylamine)	*Gammarus fasciatus*	--	S 15.5°C	48 h TL_{50}	>100	Sanders 1970
(acid)	"	--	S 15.5°C	48 h TL_{50}	3.2	"
(PGBE)	"	--	S 15.5°C	24 h TL_{50}	4.1 (2.8-5.8)	"
	"	--	S 15.5°C	48 h TL_{50}	2.6 (1.7-3.9)	"
	"	--	S 15.5°C	96 h TL_{50}	2.5 (1.7-3.7)	"
(BEE)	"	--	S 15.5°C	24 h TL_{50}	6.5 (1.0-8.6)	"
	"	--	S 15.5°C	48 h TL_{50}	5.9 (3.1-11)	"
(BEE)	*Gammarus lacustris*	~ 2 months	S 21°C	24 h TL_{50}	1.4 (1.1-1.8)	Sanders 1969
	"	~ 2 months	S 21°C	48 h TL_{50}	0.76 (0.51-1.1)	"
	"	~ 2 months	S 21°C	96 h TL_{50}	0.44 (0.31-0.62)	"
(PGBE)	"	~ 2 months	S 21°C	24 h TL_{50}	2.1 (1.7-2.5)	"
	"	~ 2 months	S 21°C	48 h TL_{50}	1.8 (1.4-2.3)	"
	"	~ 2 months	S 21°C	96 h TL_{50}	1.6 (1.2-2.1)	"
(IOE)	"	~ 2 months	S 21°C	24 h TL_{50}	6.8 (4.8-9.7)	"
	"	~ 2 months	S 21°C	48 h TL_{50}	4.6 (2.9-7.3)	"
	"	~ 2 months	S 21°C	96 h TL_{50}	2.4 (1.9-4.9)	"
(dimethylamine)	*Asellus brevicaudus*	adult	S 15.5°C	48 h TL_{50}	>100	Sanders 1969
(PGBE)	"	adult	S 15.5°C	48 h TL_{50}	2.2	"
(BEE)	"	adult	S 15.5°C	48 h TL_{50}	3.2	"

Table 7.16 (cont'd)

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
(dimethyl-amine)	Cypridopsis vidua	juvenile	S 21°C	48 h LC_{50}	8.0	Sanders 1970
(PGBE)	"	juvenile	S 21°C	48 h LC_{50}	0.32	"
(BEE)	"	juvenile	S 21°C	48 h LC_{50}	1,8	"
dicamba	Gammarus lacustris	~ 2 months	S 21°C	24 h LC_{50}	10 (6.7-15.0)	Sanders 1970
	"	~ 2 months	S 21°C	48 h LC_{50}	5.8 (3.6-9.3)	"
	"	~ 2 months	S 21°C	96 h LC_{50}	3.9 (3.1-4.9)	"
	Gammarus fasciatus	adult	S 15.5°C	48 h TL_{50}	>100	"
	Asellus brevicaudus	adult	S 15.5°C	48 h TL_{50}	>100	"
	Cypridopsis vidua	juvenile	S 21°C	48 h TL_{50}	>100	"
diquat	Hyalella azteca	4-12mm/adult	S 15°C	24 h TLm	0.58 (0.4-0.72)	Wilson & Bond 1969
	"	4-12mm/adult	S 15°C	48 h TLm	0.12 (0.10-0.16)	"
	"	4-12mm/adult	S 15°C	96 h TLm	0.048 (0.04-0.16)	"
	"	--	S 17°C	48 h LC_{50}	3.4	Williams et al. 1984
	Acellus commus	--	S 17°C	48 h LC_{50}	10	"
EPTC	Gammarus fasciatus	--	S 15.5°C	24 h TL_{50}	43 (26-75)	Sanders 1970
			S 15.5°C	48 h TL_{50}	35 (20-52)	"
			S 15.5°C	96 h TL_{50}	23 (15-36)	"
paraquat	Gammarus lacustris	~ 2 months	S 21°C	24 h LC_{50}	38 (29-49)	Sanders 1969
	"	~ 2 months	S 21°C	48 h LC_{50}	18 (12-26)	"
	"	~ 2 months	S 21°C	96 h LC_{50}	11 (8-15)	"

Table 7.16 (Cont'd)

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
picloram	*Gammarus lacustris*	2 months	S 21°C	24 h LC_{50}	50 (35-71)	Sanders 1969
	"	2 months	S 21°C	48 h LC_{50}	48 (34-76)	"
	"	2 months	S 21°C	96 h LC_{50}	27 (20-37)	"
propanil	*Gammarus fasciatus*	--	S 15.5°C	24 h TL_{50}	54 (31-82)	Sanders 1970
	"	--	S 15.5°C	48 h TL_{50}	34 (21-51)	"
	"	--	S 15.5°C	96 h TL_{50}	16 (8.5-32)	"
simazine	*Gammarus lacustris*	2 months	S 21°C	24 h LC_{50}	30 (26-35)	Sanders 1969
	"	2 months	S 21°C	48 h LC_{50}	21 (16.5-24)	"
	"	2 months	S 21°C	96 h LC_{50}	13 (11.4-15)	"
	Gammarus fasciatus	--	S 15.5°C	48 h TL_{50}	>100	Sanders 1970
	Asellus brevicaudus	adult	S 15.5°C	48 h TL_{50}	>100	"
	Cypridopsis vidua	juvenile	S 21°C	48 h TL_{50}	3.2	"
trifluralin	*Gammarus fasciatus*	--	S 15.5°C	24 h TL_{50}	3.2 (1.9-17)	Sanders 1970
	"	--	S 15.5°C	48 h TL_{50}	1.8 (1.6-12)	"
	"	--	S 15.5°C	96 h TL_{50}	1.0 (0.3-3.6)	"
	Gammarus lacustris	2 months	S 21°C	24 h LC_{50}	8.8 (6.6-12.0)	Sanders 1969
	"	2 months	S 21°C	48 h LC_{50}	5.6 (4.2-7.4)	"
	"	2 months	S 21°C	96 h LC_{50}	2.2 (1.4-3.4)	"
	Asellus brevicaudus	adult	S 15.5°C	48 h TL_{50}	2.0	Sanders 1970
	Cypridopsis vidua	juvenile	S 21°C	48 h TL_{50}	0.25	"
(Treflan)	*Cypria* sp.	--	S 20°C	48 h TL_{50}	0.06	Naqvi *et al*. 1985

[a] Formulation indicated in brackets

[b] S = static

Table 7.17. Summary of toxicity data for herbicide effects on aquatic insects.

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]	Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
atrazine	midge, *Chironomus tetans*		S 20°C	48 h LC_{50}	0.72 (0.36-1.44)	Macek *et al*. 1976
	"		S 20°C	2 generations reproductive	0.23	"
	mosquito, *Culex restuans*		S 20°C	18 h	200	Bowman *et al*. 1981
2,4-D (acid)	stonefly, *Pteronareys californica*	30-35 mm	S 15.5°C	24 h LC_{50}	56 (50-63)	Sanders & Cope 1968
	"	30-35 mm	S 15.5°C	48 h LC_{50}	44 (32-59)	"
	"	30-35 mm	S 15.5°C	96 h LC_{50}	15 (10-22)	"
(BEE)	"	30-35 mm	S 15.5°C	24 h LC_{50}	8.5 (5.7-13)	"
	"	30-35 mm	S 15.5°C	48 h LC_{50}	1.8 (1.5-13)	"
	"	30-35 mm	S 15.5°C	96 h LC_{50}	1.6 (1.3-1.9)	"
	mayfly, *Cloeon dipterum*	--	S --	48 h LC_{50}	>40	Hashimoto & Nishiuchi 1981
(dimethyl-amine)	*Ephemerella walkeri*	--	S —	lowest observed avoidance conc.	>10	Folmar 1978
(BEE)	midge, *Chironomus*	larva	S 30°C	75% mortality	1.0	Sigmon 1979
(amine)	phantom midge, *Chaoborus punctipennis*	3rd-4th instar	S 15°C	24 h IC_{50}	1490	Bunting & Robertson 1975
	"	3rd-4th instar	S 15°C	48 h IC_{50}	1142	"
	"	3rd-4th instar	S 15°C	96 h IC_{50}	890	"
	mosquito, *Aedes aegyptii*	--	S --	24 h 3.2-fold increase in mortality	10	Lichtenstein *et al*. 1973
dalapon	stonefly, *Pteronarcys californica*	30-35 mm	S 15.5°C	96 h no apparent effect	100	Sanders & Cope 1968
	mayfly, *Ephemerella walkeri*	--	S --	lowest observed avoidance conc.	>10	Folmar 1978
	dragonfly, *Aeschna* sp.	—	S —	48 h LC_{50}	>1600	Kenaga 1974
diquat	mayfly, *Callibaetis* sp.	8-12 mm	S 15°C	48 h TLm	37 (31-43)	Wilson & Bond 1969
	"	8-12 mm	S 15°C	96 h TLm	16.4 (15.0-17.8)	"
	Ephemerella walkeri	--	S 15°C	lowest observed avoidance conc.	1.0	Folmar 1978
	caddisfly, *Limnephilus* sp.	15-20 mm	S 15°C	48 h TLm	65 (61-69)	Wilson & Bond 1969
	"	15-20 mm	S 15°C	96 h TLm	33 (28-38)	"

Table 7.18. Summary of toxicity data for herbicide effects on aquatic snails and worms.

Herbicide[a]	Species Tested	Size/Stage	Test Conditions[b]		Toxic Effect Reported	Value mg L^{-1} (fiducial limits)	Reference
2,4-D (Na salt)	red snail, *Indoplanorbis exustus*	--	S	--	48 h TLm	>100	Hashimoto & Nishiuchi 1981
(acid)	marsh snail, *Semisulcospira libertina*	--	S	--	48 h TLm	>100	"
	oligochaete, *Lumbriculus variegatus*	--	S	20°C	48 h LC_{50}	122.2	Bailey & Liu 1980
	"	--	S	20°C	96 h LC_{50}	122.2	"
dalapon (Na salt)	orb snail, *Planorbis* sp.	--	S	--	no mortality	200 (a.e.)	cf. Kenaga 1974
	"	--	S	--	24 h LC_{50}	400 (a.e.)	"
	pouch snail, *Physa* sp.	--	S	--	24 h no mortality	200 (a.e.)	"
	"	--	S	--	24 h LC_{50}	400 (a.e.)	"
linuron	snail, *Limnaea stagnalis*	--	S	--	24 h LC_{50}	100	Knauf & Schulze 1972
	orb snail, *Planorbis corneus*	--	S	--	24 h LC_{100}	>70	"
	flat worm, *Dugesia tigrina*	--	S	--	24 h LC_{50}	10	"
	Planaria gocephala	--	S	--	24 h LC_{50}	10	"
	oligochaete, *Tubifex* sp.	--	S	--	24 h LC_{50}	16	"
paraquat	red snail, *Indoplanorbis exustus*	--	S	--	48 h TLm	7.2	Hashimoto & Nishiuchi 1981
	marsh snail, *Semisulcospira libertina*	--	S	--	48 h TLm	10	"
	pouch snail, *Physa acuta*	--	S	--	no effect conc.	380	Lynn 1965
picloram	snail (unknown)						
propanil	red snail, *Indoplanorbis exustus*	--	S	--	48 h TLm	14	Hashimoto & Nishiuchi 1981
	marsh snail, *Semisulcospira libertina*	--	S	--	48 h TLm	23	"
	pouch snail, *Physa acuta*	--	S	--	48 h TLm	8.8	"
simazine	red snail, *Indoplanorbis exustus*	--	S	--	48 h TLm	>100	Hashimoto & Nishiuchi 1981
	marsh snail, *Semisulcospira libertina*	--	S	--	48 h TLm	>100	"
	pouch snail, *Physa acuta*	--	S	--	48 h TLm	>100	"
trifluralin	red snail, *Indoplanorbis exustus*	--	S	--	48 h TLm	30	Hashimoto & Nishiuchi 1981
	marsh snail, *Semisulcospira libertina*	--	S	--	48 h TLm	8.0	"
	pouch snail, *Physa acuta*	--	S	--	48 h TLm	30	"

[a] Formulation in brackets

[b] S = static

to aquatic invertebrates as are insecticides (see Tables 6.1-6.6). The LC_{50} values for the most toxic herbicides (2,4-D esters, triallate, trifluralin, atrazine and simazine) are one to two orders of magnitude less than those for the same species acutely exposed to the more toxic insecticides.

Based on estimated initial concentrations (Table 7.7), only a single formulation of 2,4-D, the propylene glycol butyl ester (TGBE), produces a worst case acute exposure value which overlaps its short-term LC_{50} value. Sanders (1970) reported 50% mortality to the first instars of Daphnia magna in 48 h exposure tests at 100 µg L^{-1} 2,4-D TGBE. The 2,4-D esters should favor partitioning to biological material (Section 7.1), however the ester is rapidly hydrolyzed under most natural conditions. These opposing influences on the potential toxicity of the TGBE ester, suggest that no definitive prediction of potential impact can be made solely on laboratory data.

Although the overlap of short-term LC_{50} values and maximum expected initial concentration is a good first screen of potential invertebrate community impact, the great range of sensitivity in natural invertebrate populations and the variability in toxicity testing conditions suggest a more conservative screen for those compounds which might approach toxic levels under conditions of severe contamination. The reported toxicity values for zooplankton form the most complete herbicide data sets (Table 7.15). Only the 2,4-D TGBE, triallate and trifluralin are acutely toxic to Daphnia at levels <1 mg L^{-1} in 48 h exposures. Triallate and trifluralin have 48 h LC_{50}'s of 430 µg L^{-1} and 240-560 µg L^{-1}, respectively, for cladocerans. Naqvi et al. (1985) reported the lowest 48 h LC_{50} values for cladocerans and copepods (<50 to 80 µg L^{-1}) exposed to formulated Treflan®. As the low volatility esters of 2,4-D might be expected to reach 100-200 µg L^{-1} initially after overspray, with triallate at 270 µg L^{-1} and trifluralin at 200 µg L^{-1}, all three compounds, under these worst-case single-input conditions, are likely to cause some mortality to zooplankton.

There is also evidence of a low margin of safety between the 48 h LC_{50} values for zooplankton exposed to atrazine, simazine and amitrole and their respective calculated maximum initial concentrations. The margin is less than 10 for these herbicides (Table 7.19). Importantly, both amitrole and the triazines are expected to remain in water at high concentrations for extended periods (Section 7.1). Reproductive tests with Daphnia exposed to atrazine have demonstrated that 1 mg L^{-1} reduces the number of young per female over a 21-28 day exposure period. Life cycle tests with Daphnia exposed to picloram provided the only published estimate of the maximum acceptable toxicant concentration (11.8-18.1 mg/L) for an invertebrate species exposed to a herbicide (Gersich et al. 1985). For none of the other herbicides tested is there sufficient reason from laboratory data to consider the compound a hazard to aquatic zooplankton under acute exposure conditions if used at field recommended rates.

Positively selected duckling prey such as amphipods and isopods are sensitive to approximately the same small set of herbicides : 2,4-D, atrazine, simazine, amitrol, trifluralin and triallate (Table 7.16). Gammarids are killed by 2,4-D esters at concentrations <1 mg L^{-1} (Sanders

Table 7.19. Ratio of 48 h LC_{50} values to calculated maximum expected initial concentrations for the more toxic herbicide compounds.

Herbicide	Zooplankton	Amphipods	Insects
amitrole	2.7	—	—
atrazine	8.7	13.8	1.7
2,4-D esters	1.0	1.7	18.0
diquat	101	1.7	529
simazine	3.8	—	13.6
triallate	1.6	79.0	—
trifluralin	1.2	9.0	21.0

1969) and the 96 h LC_{50} for Gammarus fasciatus exposed to trifluralin was 1 mg L^{-1}. Atrazine is acutely toxic to amphipods at 5.7-8.8 mg L^{-1} and produced reproductive damage at 0.14 mg L^{-1} within 30 days (Macek et al. 1976). In addition, Wilson and Bond (1969) presented data which showed that the amphipod Hyallala azteca was highly sensitive to diquat (48 h LC_{50} = 0.12 mg L^{-1}); as expected, mud in test containers dramatically decreased the LC_{50} value (10.5 mg L^{-1}). The seed shrimp, Cypridopsis vidua, also was affected by 2,4-D esters and trifluralin at levels less than 300 µg L^{-1} (48 h TL_{50}) (Sanders 1970). Fish populations exposed to diquat for as little as 30 minutes have been shown to take up to a week to die (Muirhead-Thompson 1971) which suggests that diquat may have adverse effects not observable under most toxicity testing procedures. The remaining herbicides with 48 h LC_{50} values in excess of 10 mg L^{-1} are considered relatively innocuous to aquatic amphipods, isopods and ostracods.

Only atrazine has a 48 h LC_{50} for an aquatic insect species at <1 mg L^{-1} (Table 7.17): 50% of the population of the midge, Chironomous tentans, was killed at 720 µg L^{-1} (Macek et al. 1976). In a two-generation test, a concentration of 230 µg L^{-1} atrazine reduced hatching success, increased larval mortality, retarded development and reduced the rate of pupation and emergence for exposed midges (Ibid). Simazine was slightly less toxic to insects than atrazine with reported 96 and 48 h LC_{50} values of 1.9 mg L^{-1} and 3.58 mg L^{-1} for stoneflies and midges, respectively. The esters of 2,4-D also caused mortality in exposed stoneflies (Sanders & Cope 1968) and midges (Sigmon 1979) at low concentrations. The 96 h LC_{50} for trifluralin-exposed stoneflies was 3.0 mg L^{-1}. Other herbicides should not pose a significant hazard to aquatic insects.

Laboratory toxicity test data for aquatic snails and worms suggest that these taxonomic groups are not highly susceptible to direct herbicide toxicity (Table 7.18). Only for propanil and trifluralin were 48 h LC_{50} values less than 10 mg L^{-1}. However, snails in natural slough habitat may, because of feeding habits, be exposed to higher levels of the herbicide than what is available in water. This exposure pathway is examined more fully in the field studies section which follows.

Table 7.19 summarizes the relative potential hazard posed by the most toxic herbicides under a "worst-case" situation of maximum exposure from an aerial overspray of a shallow pond. A ratio of 1 indicates equal 48 h LC_{50} and calculated maximum initial concentration values. Ratio values <10, and especially those approaching 1, suggest potential danger to highly sensitive species. A need to examine chronic effects of the more toxic and persistent herbicides on critical prey species then exists.

As a group, zooplankton appear most susceptible to herbicide toxicity. However, as a prey group they are of minor importance in the diet of ducks, with the exception of Shovelers. Amphipods are at great risk to diquat toxicity in its non-adsorbed state. Because of their habits, amphipods may be further exposed to relatively high and stable diquat concentrations while feeding on contaminated sediments and aquatic plants. Insects as a group were most sensitive to the triazine herbicides, particularly atrazine, which has demonstrated chronic effects on midges. Chronic studies are not available for trifluralin, although it has been shown to persist under aerobic conditions and can be bioaccumulated and magnified to some extent in the food chain (Metcalf & Sanborn 1975). The 2,4-D esters, atrazine, simazine and trifluralin, can pose a consistent and moderate hazard to the major invertebrate taxa contibuting substantially to duckling feeding. The severity of hazard they pose to the aquatic invertebrate community, based on the analysis of laboratory direct toxicity data, is relatively minor in comparison to that posed by the highly toxic insecticides.

7.3.1.2 Field studies

There is little evidence in the literature that applications of herbicides to shallow lentic water bodies, at normal crop protection rates, have direct toxic effects on aquatic invertebrate populations. Multiple treatments of 1 mg L^{-1} diquat reduced the number of cladocerans in dugout ponds (Gilderhus 1967). Way et al. (1971) reported that paraquat treatment (0.5 mg L^{-1}) of a shallow lake probably eliminated Gammarus through direct toxic mortality or oxygen deprivation. Their conclusion was based on the 20% mortality of Asellus held in traps in the treated lake. Clams exposed to atrazine at 0.5 mg L^{-1} in a pond enclosure were reduced to 10% of original numbers but snail populations increased to nearly four times pretreatment levels (Walker 1962). The numbers of Daphnia in pond enclosures treated with linuron at 1 mg L^{-1} declined rapidly and did not repopulate within 35 days (Stephenson & Kane 1984). Linuron also reduced arthropod species richness and population abundance.

There are a greater number of reports of impacts on aquatic invertebrates resulting from the herbicide destruction of macrophytes. The stems and leaves of submergent and emergent plants provide an important habitat for snails, amphipods, midges, dragonflies, mayflies, caddisflies, aquatic moths and large zooplankton. The macrophytes provide substrate, food and protection from predators for these weed-clinging species. In reviews of herbicide impacts in aquatic systems, both Newbold (1975) and Brooker and Edwards (1975) indicated that the primary impacts on invertebrate fauna were associated with the loss of macrophyte habitat. The general pattern was an elimination of species having an obligate association with these aquatic plants as the dead macrophytes decompose and are no longer available as substrate. There was a parallel increase in detritus feeders such as oligochaete worms. Zooplankton may also be indirectly affected as increased light and availability of nutrients from macrophyte die-back stimulate phytoplankton growth and subsequently the productivity of open water zooplankton.

Hilsenhoff (1966) reported that a 1 mg L^{-1} diquat addition to a pond led to greatly reduced numbers of snails (Helisoma, Gyralus, Physa) and amphipods (Hyalella) associated with the elimination of the water weed Elodea within seven days. With the virtual control of Hydrilla after diquat and copper treatment (1 mg L^{-1} diquat), chironomid midge larvae declined from 2,200 m^{-3} to 700 m^{-3}; crustaceans and caddisflies also exhibited population declines (May et al. 1973). Paraquat has been shown to produce similar indirect changes in invertebrate fauna. Chironomids (Orthocladiinae), tricopterans (particularly Agraylea multipunctata), lepidopterans (Litodactylus leucogaster and Acentropus nivens) and snails (especially Planorbis laevis) were markedly reduced in a paraquat-treated reservoir following the destruction of angiosperms (Brooker & Edwards 1974). Acentropus larvae associated with Potamogeton declined from a pre-treatment peak of 950 m^2 to 0 m^2 and did not recolonize the next year. As in the case of Acentropus, habitat destruction during the periods of egg development and/or shortly after hatch, is particularly detrimental to species re-establishment. Two to four weeks post-treatment, large cladocerans, snails, some dipteran and lepidopteran larvae were absent from a 2,4-D DMA-treated reservoir mainly due to the absence of Eurasion water milfoil, Myriophyllum spicatum (Wojtalik et al. 1971). In a similar reservoir treatment, 2,4-D BEE indirectly eliminated elmid beetles, leptocerid caddisflies, the mayfly Caenis, and most dragonfly naiads (Smith & Isom 1967). Walker (1963) reported that 1 to 2 mg L^{-1} monuron reduced the production of bottom fauna by 20% three months after treatment. The species composition reflected the change in habitat: the weed-clinging species of Gastropoda, Odonata, Tricoptera and Ephemeroptera were eliminated. Scorgie (1980) reported the near eradication of the weed-clinging snail Lymnaea peregra with the application of 0.2 mg L^{-1} cyanatryn to clear water milfoil from drainage channels.

Walker (1971) suggested that herbicides reroute the energy flow from primary consumers to higher level feeders such as fish. His conclusions were based on the indirect impact of 0.5 mg L^{-1} sodium endothal on the fauna of a small pond through the decay of the major macrophytes. The importance of snails, dragonflies, damselflies and caddisflies in the food web was reduced with herbicide treatment. Oligochaetes and midge larvae increased in importance as fish food.

The indirect effects of herbicides on zooplankton were reported by Walsh et al. (1971), Newbold (1974) and deNoyelles et al. (1982). Species associated with vegetation or murky water such as Simocephalus vertulus were replaced by clear water species like Daphnia longespina following herbicide treatment, macrophyte decay and the transition to a more plankton-dominated system (Newbold 1974). The abundance of open-water species may increase with the "bloom" of phytoplankton. Possible secondary impacts of pesticides on both macroinvertebrates and zooplankton were thoroughly reviewed by Hurlbert (1975).

Factors which affect the seriousness of elimination (or reduction) of weed-associated species as waterfowl food resources are 1) the extent of macrophyte destruction, 2) the rapidity of re-vegetation, 3) the availability of alternative substrates for macrophyte clingers, and 4) the importance of these species as duck food. Newbold (1975) points out that the ecological damage to pond invertebrates by the removal of dominant macrophytes may be long-lasting if 1) the herbicide is non-selective, 2) the whole pond is sprayed (no source of recolonizers), and 3) spraying is carried out in consecutive growing seasons. Adults and ducklings of dabbler species like the Mallard are known to feed around aquatic vegetation and the work of Armstrong and Nudds (1985) pointed out the importance of the macrophyte habitat as a source of large prey items. A reduction in snail populations associated with macrophytes could be damaging if it occurs while the hen is developing eggs (Section 2.5). Gastropods also provide between 4-10% of the diet of young Pintails and Scaup (Ibid). Chironomid larvae are important in the diet of many young ducklings, as are gammarids like

Hyalella. Damselflies, dragonflies and caddisflies may provide up to 5% of gross energy to young ducklings (Ibid). Individuals of these insect species are relatively large in size and represent a large number of calories per catch and thereby an important energy source. However, as there is no data indicating obligate duck feeding relationships with weed-clinging invertebrate species, the importance of their elimination from herbicide-treated ponds cannot presently be adequately evaluated, with the possible exception of snails as cited above.

It is difficult to assess in quantitative terms what proportion of total macroinvertebrate abundance in sloughs is solely dependent upon macrophytes. Joyner (1982) found that the average number of macroinvertebrates in ponds associated with vegetation was approximately ten times the abundance in the bottom sediments or the water column. Mittelbach (1981) reported a greater biomass of invertebrates associated with aquatic vegetation than in the benthic sediments. This fact should not be surprising as it has been estimated that the surface area provided by macrophytes may be 50 times greater than the surface area of the pond bottom sediments (from Hurlbert 1975). However, some invertebrates closely associated with macrophytes prior to herbicide treatment are able to colonize other available substrates upon loss of the angiosperms. Brooker and Edwards (1974) reported that the macroalga Chara grew rapidly after paraquat treatment and provided suitable substrate for the water mite Hydrozetes confervae. Chara, as well as vegetation at the water's edge unharmed by paraquat, also provided refuge for the corixid Sigara. Most herbicide impact studies, unlike that of Brooker and Edwards (1974), have not attempted to adequately sample the macrophyte zone for invertebrates and have paid little attention to either changes in species composition or total abundance of this important component of the slough system. If herbicide overspray or runoff in prairie potholes substantially reduces macrophytes, and if only 30% of the weed-inhabiting individuals were lost, this would still represent approximately 28% of the total macroinvertebrate community, based on data from Joyner (1982) and Mittelbach (1981). Therefore, the impact of herbicides on invertebrate food resources for ducks depends on the 1) the severity of macrophyte kill, 2) the particular taxa and the total number of macrophytedependent individuals eliminated, and 3) the behavioural changes in feeding strategy of the duck species which feed within emergent and submergent macrophytes. At present, there is insufficient information specific to the pothole region on all of the above aspects of the hazard assessment problem.

7.3.2 Herbicide impact on plant foods

Section 2.5 of this report clearly identified the invertebrate food of ducks as a key element in the successful production of waterfowl in the prairies. Laying females and newly hatched ducklings are heavily dependent on animal foods for egg production and growth, respectively. At other times of the year, or of their life-cycle, many duck species depend to some extent on plant foods. The following section will discuss the importance of plants as food for ducks and the possible impact of herbicides on pothole vegetation and the subsequent effect on waterfowl.

The plant foods of ducks are of four types: the seeds (or fruits), the tubers, the foliage and the reproductive structures. While the seeds of numerous species found in potholes are consumed by ducks, the other plant parts are subject to greater selection. The tubers of the pondweed, Potamogeton, are particularly choice foods while foliage is mostly taken from duckweed (Lemna), the macroalga Chara and the pondweed Potamogeton. The reproductive structures or oögonia of Chara are also chosen. Table 7.20 lists those foods which contributed to more than 1% of total foods as indicated by the analyses of 399 stomachs from 13 species of ducks from western Canada. The six plant species listed in this table are all important components of pothole vegetation.

The plant foods found in and around potholes, which are selected as food by 11 species of ducks are presented in Table 7.21. This list is not exhaustive; it gives an overview of the plants. There is a large amount of overlap between ducks in the plants consumed: certain foods such as pondweeds and seeds of bulrushes are found in almost all diets. The foods chosen are varied and indicate a lack of specialization. While most of the birds prefer seeds, three species, the Blue-winged Teal, the American Widgeon and the Gadwall, consume mostly foliage. In contrast, Shovelers and Lesser Scaup have diets consisting mostly of invertebrates. Of all the duck species, the Mallard and Pintail, and to a lesser extent the American Widgeon and Green-winged Teal, have adapted well to using grain crops as food.

In examining Table 7.21, one must keep in mind that duck diets are highly site-specific (see Bellrose 1976): plant foods in the prairies are different than those in the wintering sites on the coasts or in the southern United States. The ratio of animal to plants in the diet is also highly variable. In Chesapeake Bay, the decline of beds of aquatic plants in recent years has led to a switch from eelgrass and widgeon grass to clams as the major item in the diet of the Canvasback (Bellrose 1976). Seasonal changes in diet composition have also been noted, as both the invertebrate and plant components of the diet are strongly influenced by availability (Perret 1962, Dirschl 1969, Serie & Swanson 1976).

As was discussed in Section 7.2, the effect of the use of herbicides in the prairies on pothole vegetation cannot be predicted from the information available. While certain compounds such as the triazines, 2,4-D amine, diquat, paraquat and glyphosate were identified as potentially the most damaging, the factors mediating their toxicity to wetland vegetation (e.g. mode of exposure, dose-response, timing of application) have been largely left unexamined. Therefore, one can only speculate as to the impact on waterfowl plant foods and its significance to the duck populations.

Plants are more susceptible to the action of herbicides during the early stages of growth. Exposure to herbicides early in the season may not only prevent further development of the vegetation through the summer, but may also remove, as a source of food, the tender leaves and stems of young shoots which would be choice foods at that time of year. During most of the summer, the ducks rely to a great extent on the abundant invertebrates of prairie potholes. Herbicide reduction of plant biomass and/or production may have little direct impact on duck feeding during the summer. However,

Table 7.20. Foods of ducks in western Canada[a].

Scientific Name	Common Name	Volumetric Percentage	Parts Selected			
			Seeds	Tubers	Foliage	Oögonia
Potamogeton	pondweed	12.73	X	X	X	
Scirpus	bulrush	10.58	X			
Chara	stonewort	9.29			X	X
Polygonum	smartweed	2.49	X			
Myriophyllum	water milfoil	2.28	X			
Carex	sedge	1.50	X			

[a] Modified from Martin and Uhler 1939

Table 7.21. Predominant plant foods consumed by 11 species of waterfowl in the prairies[a].

Species	Seeds	Foliage	Tubers	Oögonia	Invertebrates[b]	Crop Grain[c]
Mallard	bulrushes pondweed spike rush millet smartweed	pondweed widgeon grass coontail			X	X
Pintail	bulrushes pondweed	*Chara* smartweed pondweed widgeon grass		*Chara*	X	X
Blue-winged Teal	sedge bulrush	filamentous algae *Chara* duckweed widgeon grass coontail pondweed			X	
Northern Shoveler	bulrush pondweed				XX	
American Widgeon		pondweed coontail wild celery algae filamentous algae			X	X
Gadwall		pondweed widgeon grass coontail *Chara* filamentous algae			X	

Table 7.21 (cont'd)

Species	Seeds	Foliage	Tubers	Oögonia	Invertebrates[b]	Crop Grain[c]
Green-winged Teal	millet smartweed sedges bulrush pondweed spikerush	*Chara* pondweed widgeon grass duckweed			X	X
Lesser Scaup	bulrush pondweed water lily	pondweed widgeon grass *Chara*			XX	
Redhead	pondweed bulrush water lily	pondweed filamentous algae *Chara* wild celery duckweed	pondweed	*Chara*	X	
Canvasback	pondweed bulrush	*Chara* wild celery widgeon grass	pondweed	*Chara*	X	
Ruddy Duck	pondweed bulrush	pondweed widgeon grass *Chara*	pondweed	*Chara*	X	

[a] Information gathered from Bellrose 1976

[b] Entries in this column represent some consumption of invertebrates (X) or a very high contribution of invertebrates in the diet (XX)

[c] An entry in this column indicates that the species has adapted well to using grain crops for food

if herbicides reduce production or even prevent flowering of wetland vegetation, storage of energy reserves in the form of tubers and seed production may be severly reduced. Tubers and seeds are important sources of food for ducks in late summer and early fall when invertebrates are less abundant because they allow the ducks to build up reserves before the fall migration. Also, seeds are an important source of food for birds arriving early in spring on the breeding grounds when invertebrates and other plant foods are scarce. Any reduction in seed numbers through exposure to herbicides the previous summer may reduce the chance of survival of ducks arriving early when weather conditions are still harsh.

The species most susceptible to decreases in plant foods may be the Gadwall and American Widgeon as they rely most heavily on plant foods. Also, the ducklings of these two species switch to a diet consisting almost entirely of plant foods within three weeks after hatching (Sugden 1973).

Whereas the availability of abundant invertebrate prey is crucial for successful brood production and duckling growth, and can be limiting under certain circumstances, plants, as food, have not been shown to be a factor limiting populations of ducks. The ability of ducks to feed opportunistically on cereal grains or to switch from a plant-dominated diet to one consisting mostly of invertebrates shows a high degree of adaptability to local food conditions. The mobility of the adults also allows them to forage over wide areas. These factors, along with the variety of plant species and plant parts used as food, suggest that ducks are highly flexible when confronted with either natural or herbicide-induced food resource fluctuations.

Other factors which may minimize the effects of herbicides in sloughs are the very high primary productivity of these systems and their ability to recover relatively rapidly from herbicide exposure. As was discussed in Section 7.2, herbicides often do not reduce production but instead lead to changes in dominants: the more sensitive species are replaced by more resistant forms or by species which recover more quickly such as Chara and filamentous algae, which are both duck food. In addition, most plant species affected by herbicides recover within one year (Table 7.14). Although leaves and stems die off after exposure to herbicides, tubers and seeds are unaffected.

In summary, it seems likely that the herbicide impact on plants will be directed to the ducks through pathways other than as a source of food. First, through the elimination of invertebrates via the destruction of the habitat provided by the vegetation (Section 7.3.1). Second, through the elimination of nesting and brood-rearing habitat as will be discussed in the following section.

While it is clear that the preceding evaluation of the impact of herbicides on duck plant foods is based mostly on informed guesses, a proper assessment can be made only after more work is done to answer two major questions: (1) how will herbicide use affect slough vegetation, and (2) how do ducks respond to fluctuations in plant food resources?

7.4 Herbicide Damage to Nesting Habitat

The importance of plant cover to breeding ducks has been clearly outlined in Section 2.5 of this report. Both the amount and quality of vegetative cover available in and around potholes contribute greatly to the success of nesting hens and the survival of broods. Plant cover provides concealment from predators for nests and for broods moving between potholes. Also, marsh vegetation provides shelter from harsh weather conditions to which ducklings, especially those of early nesting species, are susceptible.

The habitats of greatest importance for breeding ducks are the uplands and the potholes and their margins. When undisturbed, these provide good cover. In the uplands, snowberry, willows, woodrose, clover, nettles, thistles and mixed prairie grasses have all been shown to provide favored nesting cover for the hens. In and around potholes, cattails, sedges, bulrushes, whitetop and reeds are important components of good brood-rearing habitat. Within agricultural areas in the prairies, however, little upland vegetation is left undisturbed. The habitats which still provide good cover consist mostly of roadsides, slough margins, rights-of-way, hay fields and idle land (see Table 2.25). Because of their small size and isolation, the maintenance of abundant plant cover within these habitats is of great importance to duck production.

The impact of herbicide use, in the prairies, on such habitats has not been investigated. While herbicide effects on weed populations in cultivated fields is relatively well researched, their impact on natural plant communities in the prairies is unknown. However, there have been a number of studies of natural plant assemblages (i.e. non-crop) sprayed with herbicides for management purposes (e.g. forests, roadsides, utility rights-of-way). We will attempt, therefore, to summarize these studies and extrapolate to a prairie situation.

While the initial effect of herbicide application is a reduction in abundance of the undesirable species, the results is often variable and depends on the original species composition of the habitat, the type of plant targeted, the type of herbicide applied and the frequency and selectivity of its application. For example, in the control of woody vegetation along utility, railroad and roadside rights-of-way, blanket spraying of phenoxy compounds leads to the establishment of spray-resistant grasses which are unstable and allow the re-establishment of trees and shrubs; thus, continual treatment is required (see Niering & Goodwin 1974). On the other hand, if trees are sprayed selectively, relatively stable shrublands are created which resist reinvasion by nearby trees (Bramble & Byrnes 1972, Niering & Goodwin 1974).

Due to variations in the susceptibility of plants to herbicides, the use of the more selective compounds will most often lead to interspecific shifts in populations (see NRCC 1978b). Yemm and Willis (1962) found that the use of maleic hydrazide on roadside vegetation over a period of three to four years led to the suppression of growth and flowering of grasses which were replaced by rhizomatous grasses and dicots. The use of picloram to control aspen in Alberta had detrimental effects on the understory of Rosacea, Leguminosae and Compositae; grasses subsequently invaded the areas previously dominated by forbs and shrubs (Kroeger & Shaw 1982). Changes in plant community composition are not always accompanied by changes in plant biomass. Application of 2,4-D on an alpine plant community did not appreciably change the total standing crop, although the graminoid to forb ratio shifted from 3:7 to 8:2 (Thilenius et al. 1974). In conjunction with shifts in species composition, herbicides lead to changes in the physical structure of the habitat: the use of 2,4-D for 12 years on roadsides caused a decrease in average plant height from 90-95 cm to 50 cm along with a shift to dominance by grasses (Willis 1972).

Herbicides also reduce plant species diversity. Willis (1972) found that the number of plant species declined by more than 50% on roadsides sprayed annually with 2,4-D for 12 years. The herbicides 2,4-D, picloram, diuron, simazine and paraquat all reduced species diversity in old-field communities (Tomkins & Grant 1974, 1977). Each herbicide affected the community differently: the auxins led to the proliferation of monocots, simazine reduced monocots, paraquat eliminated both monocots and dicots and diuron simplified the community without changing the forb to grass ratio. While all herbicides simplified the community, the duration of the effect varied markedly with the compound used: vegetation treated with auxins, paraquat and simazine recovered rapidly, whereas diuron had long-lasting effects.

The response of plants to herbicides also depends on the type of community in which they are growing. Tomkins and Grant (1974, 1977) found that weeds established in disturbed areas (younger plant community) were more susceptible to herbicides than most of those growing in a more mature community. Non-selective herbicides were also found to affect more severely the younger plant community in an old-field ecosystem. Kroeger and Shaw (1982) found that when picloram was applied to control aspen, forbs and shrubs in long-established aspen forest were affected but those on dry grassland sites were unscathed.

Such changes in species composition brought about by herbicide use are likely to affect a variety of wildlife species which use the plants as nesting sites or as food (either by the direct consumption of plant parts or by consumption of the invertebrates associated with plants). The loss of some plant species in favor of others should similarly lead to the decrease of certain species of wildlife and the concomitant increase in abundance of others better suited to the changed habitat. For example, the stable shrub community which became established on a sprayed utility right-of-way in Pennsylvania was heavily used by deer, rabbits, grouse and wild turkeys which were absent in the adjoining upland oak forest (Bramble & Byrnes 1972). Kroeger and Shaw (1982) noted that after aspen was killed with glyphosate, passerine bird activity decreased but small mammals were unaffected. Changes in brush species composition in a jeffrey pine plantation after treatment with 2,4,5-T resulted in substantial reductions in both the number of resident bird species and in the size of the populations; at the same time mule deer were reduced whereas some small mammals increased (Savidge 1978). Slagsvold (1977) found that spraying a clear-felled spruce forest with 2,4,5-T reduced the number of bird territories by 30%. This, he postulated, reflected the lower bird density and diversity of early successional stages of plant communities.

These studies differ somewhat from the findings of Morrison and Meslow (1984) who looked at the effects of glyphosate on bird community structure in clearcuts in Oregon. They found that while the species composition of the birds was modified by the spraying, there was no difference in bird densities between the sprayed and control plots.

Predicting changes in wildlife populations after herbicide use requires a clear understanding of the use they make of the various plant species and of the flexibility in that use. When brush fields were treated with 2,4,5-T, two species of shrubs were killed but there was no effect on bird species composition, relative abundance or population size (Beaver 1976). Although the two species of shrubs were killed, they kept their leaves for as long as two years and still provided adequate cover for nesting. In plots where sagebrush was eliminated by 2,4-D (Best 1972), Brewers Sparrow, which nest in shrubs, declined whereas the population of Vesper Sparrows, which nest on the ground, was unaffected; also of interest, plant foods (grass seeds) became a greater component of the diets of both species. When glyphosate was sprayed on clearcuts in Oregon, some bird species adapted to the changed conditions by switching from using shrubs to using trees (Morrison & Meslow 1984).

Of particular interest to this report are some studies done on the impact of pesticide use in cereal crops on populations of Grey Partridges in England. Rands (1985) demonstrated how cereal field headlands left unsprayed with herbicides and fungicides had greater Grey Partridge brood sizes than sprayed headlands. Partridge chick survival was strongly dependent on invertebrate abundance. Herbicide treatment for dicots control reduced arthropod abundance by more than two-thirds in a similar experiment (Southwood & Cross 1969) in cereal fields. Herbicides reduced invertebrates through both direct toxicity and the elimination of host plants.

It is clear from the studies described above that the changes in plant communities brought about by herbicides, and the subsequent responses of wildlife, are variable. The prediction of such effects requires a knowledge of the composition of the plant community before spraying, the type of herbicide used, the frequency and selectivity of application and the dependence of wildlife species on their habitat. In the Canadian prairies, the composition of plant communities in those habitats which still provide attractive nesting cover for ducks is variable. These habitats differ substantially from one another (e.g. roadsides vs. idle land) and are affected by numerous agricultural activities (e.g. grazing, burning, mowing). Without specific studies on the effect of herbicide application on such habitats, it is not possible to predict the effects on vegetation and nesting ducks. There is, however, one study which may provide insight into the consequences of the contamination of nesting habitat by herbicides. The effects of herbicide sprays on the vegetation and nesting ducks on two islands in Miquelon Lake, Alberta, were studied by Dwernychuk and Boag (1973). The pre-treatment plant community consisted of luxuriant stands of thistles, nettles and mustard along with grasses and, in low-lying portions, bulrushes and sedges. The early-nesting species of ducks (Mallard, Pintail and American Widgeon) favored dead perennial forbs from the previous year as nest sites while the late-nesting ducks (Lesser Scaup, Gadwall and White-winged Scoter) selected new growths of sedges and nettles.

When plots were sprayed with 2,4-D (2.24 kg/ha) in July 1966, flowering of broad-leaved plants was prevented and by August the coverage of broad-leafs was reduced by 85% relative to unsprayed zones. In the spring of the following year the amount of residual plant cover was severely diminished (cover of thistle reduced by 92%) and, one year after treatment, grasses had increased by 219%. Close inspection of the nesting data shows that nest density was reduced by 74% in plots sprayed with 2,4-D. When herbicide treatment was stopped, an increase in the number of nesting ducks was correlated with a 300% increase in the size of area dominated by plant species preferred by ducks. Nest distribution data indicate a strong preference by ducks for specific plants and the authors therefore concluded that the decrease in nesting ducks resulted from the herbicide-induced decline in the preferred plants.

The study area used by Dwernychuk and Boag (1973) is not representative of duck nesting habitats in the agricultural landscape of the prairies. There was no disruption of the habitat by agricultural practices on the islands, nest densities were as much as two orders of magnitude greater than in and around cultivated lands and there were probably few, if any, terrestrial predators. Nevertheless, the study did demonstrate clear links between herbicide use, decreased plant cover and numbers of nesting ducks. While one may not be able to quantitatively predict the effects of herbicide use in the Canadian prairies on ducks and their habitat, the potential for loss of plant cover and, subsequently, a drop in the number of nesting ducks is clearly present.

CHAPTER 8

HAZARD EVALUATION

8.1 Introduction

Approximately 26% of the total North American duck population (or about 16 million breeding birds) arrive on the Canadian prairies each spring to reproduce. Of those, slightly more than 10 million nest in intimate association with agricultural cropland, either in the crops proper or, more commonly, in field edges, slough borders, along fence lines, etc. The remaining six million nest in pastures and other land of lesser agricultural quality or in the few remnants of natural habitat scattered throughout the prairies. Due to agricultural encroachment, suitable nesting and feeding habitat is disappearing at an alarming rate. As a result, the seemingly inexhaustible supply of sloughs is dwindling and the ability of the remaining ones to support waterfowl is at risk. The numbers of some of our most valuable harvested species (as well as unknown numbers of other species for which we do not have enough information to make an assessment) are steadily declining. One of the initial steps in protecting these species is to develop an understanding of the major factors involved in their decline.

One factor which has been almost totally ignored by the scientific community as well as by the resource managers, is pesticide use. Pesticides are a ubiquitous factor in today's agricultural landscape and their characteristics are such that they have the potential of having both a direct and indirect (or ecologically mediated) impact on the waterfowl resource, although this impact, in most cases, is not as readily visible as other forms of impact such as drainage. In Chapters 2 to 7 the potential for pesticide impact was outlined. What we have not yet determined is the extent to which these impacts are being realized today.

The purpose of this chapter is to assess the hazard posed to waterfowl populations by pesticide use in the prairies. In this context, the term hazard signifies the probability of exposure to pesticide contamination and the consequences of such an exposure. Depending on the case developed, both the exposure and the consequences will vary in their form. For example, exposure may refer to the contamination of the ducks or of the habitat while the consequences may be either mortality through direct toxicity, starvation or predation due to decreased habitat quality.

Using information provided in previous chapters, the number of ducks which may be affected by pesticide use is determined under various conditions (i.e. type of compound used, crop and habitat type). The consequences of scenarios of contamination of given numbers of ducks or areas of duck habitat are then outlined. For certain cases of pesticide use the consequences will be clearer and more direct than for others. Exact figures on mortality or on the extent of the decline in productivity caused by pesticides cannot be given for a number of reasons:

1) within the two groups of pesticides (insecticides and herbicides), there are over 50 different compounds used in the prairies. Our knowledge of actual pesticide use is very incomplete (Chapter 4);

2) pesticides can affect waterfowl through three main pathways: direct acute toxicity, nutritional deficiency of ducklings and laying females due to the mortality of invertebrate populations, and reduced nesting density and nesting success due to deterioration of vegetative habitat;

3) many other agricultural practices, such as plowing, mowing, sowing, burning, harvesting, draining, etc., can also affect duck survival and productivity. It is almost impossible to isolate the impact of pesticide use from the unquantified impacts of these other agricultural practices;

4) what little data are available, both on duck productivity as well as on agricultural practices in the prairies, are highly variable and often unreliable. In Chapter 3 we cautioned the reader about the extent of manipulation of the population data. The same warnings are in effect here since much of what follows is based on those estimates. Although more survey information is available for any one waterfowl species than for all non-game bird species put together (let alone mammals, amphibians or reptiles), the scale at which this information was collected is still inadequate for our needs.

Due to major differences in their use patterns and effects, insecticides and herbicides are evaluated separately using different approaches. Herbicides are applied yearly over a wide area of the prairies. This allows a relatively simple approximation of the number of birds exposed. A theoretical baseline production of broods is calculated for the 10 million ducks nesting within the agricultural habitat of the prairies assuming that all herbicide use is on the crops proper and that non-crop areas are completely free of any herbicide impact. The effects of chronic exposure to herbicides on the quality of margin nesting habitat can be assessed against this baseline.

In contrast, insecticide use is patchy and variable in space and time and with respect to the volumes applied because it reflects pest outbreaks which are also variable. For these reasons, three different scenarios of exposure and effects are developed which represent an approximate baseline use as well as two "outbreak" situations for the prairies: a major grasshopper outbreak and an outbreak of a major canola pest. In each case the number of birds and/or broods exposed is determined and the consequences specific to the crop sprayed, the method of application and the compound used are outlined.

In the final analysis, some of the questions we raise cannot be answered by a report such as this one no matter how complex or complete it is. Rather, some relatively simple and straightforward field investigations are necessary to provide the answers. In the absence of such investigations, we offer this analysis hoping that it will spark the interest of resource managers and will encourage some basic research aimed at looking at the impact of pesticides on waterfowl and other wildlife populations.

8.2 Herbicides

8.2.1 Hazard scenarios - input

In the grassland-parkland region of the Canadian prairies about 310,315 km^2 of land are under crop or summer fallow. From early spring to early fall, approximately 10.37 million ducks (calculated from a 10-year average of population estimates; see Chapter 3) make use of this area for breeding. During the same period, approximately 131,780 and 102,950 km^2 of the area are sprayed annually with broadleaf and grass herbicides, respectively (see Section 4.3). Using data on the distribution of dabblers by census district, crop and habitat type (Appendix 3.9), and the proportion of crops sprayed (Table 8.1), it is estimated that over five million ducks, or approximately 50% of the waterfowl in the prairie grassland-parkland region, are found in areas where herbicides are used on an annual basis.

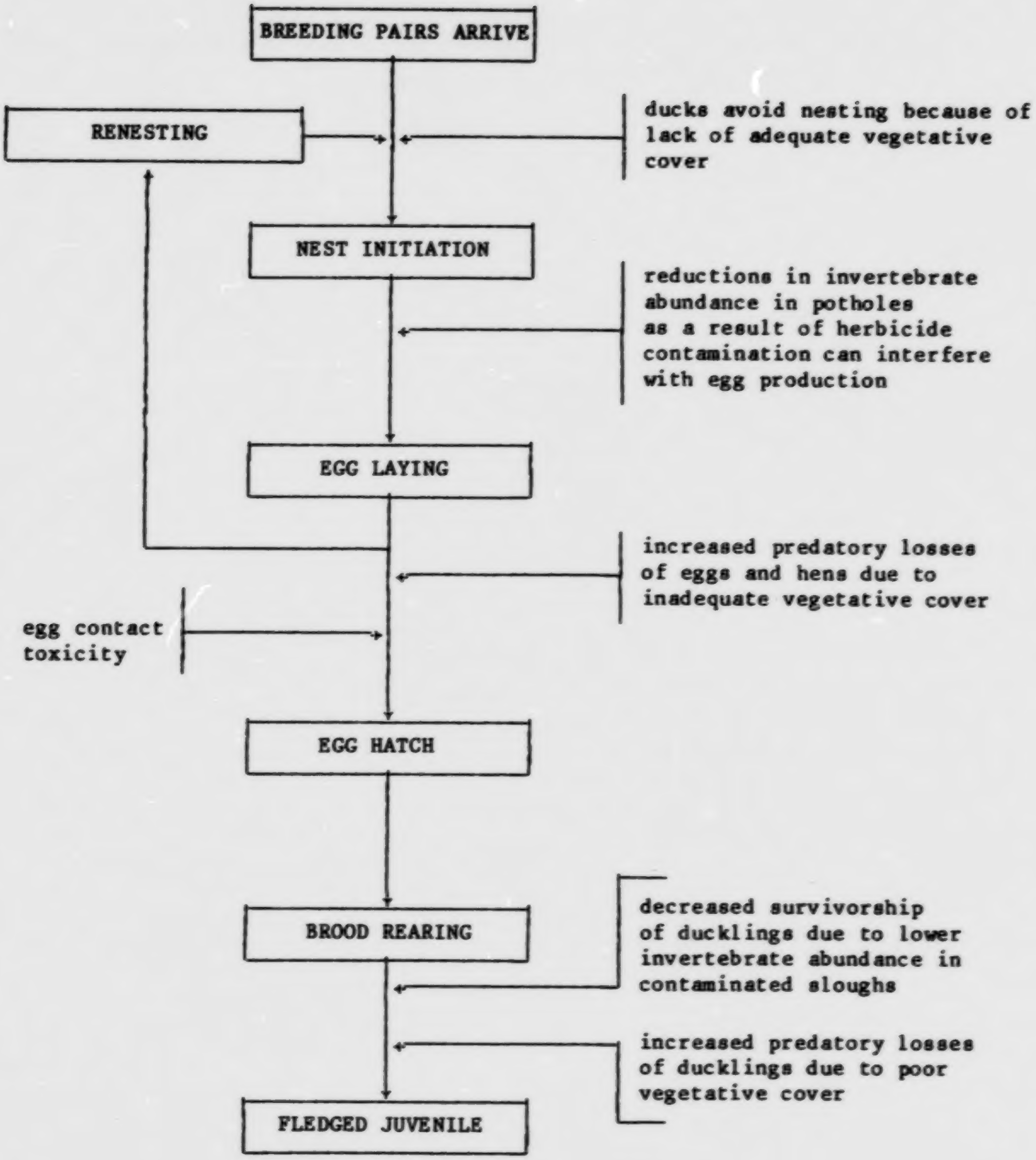
The impact of agriculture on ducks and duck habitat originates from numerous activities. Studies available on duck breeding success indicate lower recruitment rates in agricultural areas than in more pristine areas (see Section 2.5). The individual factors contributing to such a decrease in breeding success have been identified but rarely weighed. Farmers plow, till, sow, burn, mow, harvest and drain the fields and also occasionally add fertilizers, herbicides and insecticides. Any one, and sometimes all, of these factors interfere with various phases of the duck breeding process. It is, therefore, almost impossible to isolate herbicide use from the other activities contributing to decreased breeding success.

Within the cultivated field proper, it is clear that so many activities take place which are destructive to duck nests and habitat that herbicide sprays will probably add little to the disruption of the natural system already taking place. By nature, the cultivated field is a highly simplified environment (i.e. monoculture) maintained that way for one purpose: to grow a crop with the highest possible yield.

In spite of these poor conditions, over 10 million ducks still breed within the agricultural landscape. The two major habitat components required for successful breeding are (1) a complex of shallow bodies of water for feeding and (2) adequate vegetative cover near the potholes for nesting (see Section 2.5). Within the agricultural landscape the best vegetative cover is found mostly in idle grassland, hay fields, rights-of-way, pothole margins, roadsides or headlands. These areas usually account for less than 10% of the agricultural land (Higgins 1977, Sugden & Beyersbergen 1984) and are interspersed throughout the landscape. The prime nesting habitat is, therefore, rare and spread out between large areas of poor habitat. This assessment will focus on herbicide contamination of those habitats which still provide conditions suitable for successful reproduction of ducks.

Figure 8.1 illustrates various mechanisms by which herbicides can interfere with the duck breeding cycle. Three basic modes of action underlie the impact of herbicides: (1) through a reduction in invertebrate abundance in contaminated potholes, (2) through egg contact toxicity and (3) through a decrease in vegetation necessary to protect eggs, incubating

Figure 8.1. Possible points of impact of herbicides on ducks during their breeding cycle.


BREEDING PAIRS ARRIVE
RENESTING
ducks avoid nesting because of lack of adequate vegetative cover
NEST INITIATION
reductions in invertebrate abundance in potholes as a result of herbicide contamination can interfere with egg production
EGG LAYING
increased predatory losses of eggs and hens due to inadequate vegetative cover
egg contact toxicity
EGG HATCH
BROOD REARING
decreased survivorship of ducklings due to lower invertebrate abundance in contaminated sloughs
increased predatory losses of ducklings due to poor vegetative cover
FLEDGED JUVENILE

hens and ducklings against predators. Considering the first means of impact, there is little evidence that, at the rates applied in the prairies, the herbicides will cause invertebrate die-offs in potholes (see Section 7.3). Atrazine, simazine, trifluralin, triallate and esters of 2,4-D pose a moderate hazard to the invertebrate taxa contributing the most to duckling feeding, however, these compounds are still two to three orders of magnitude less toxic than the most toxic insecticides. Laboratory and pond studies suggest that decreases in invertebrate abundance would more likely be the result of the destruction of macrophyte beds than due to direct acute toxicity of most herbicides (see Chapter 7). However, this avenue has not been explored experimentally in prairie potholes and therefore does not permit the development of scenarios of herbicide impact via the food supply.

We have no means by which to assess the field relevance of the egg contact toxicity data presented in Appendix 1.1. The use of trifluralin at least should be carefully examined, especially in canola growing areas where it is used extensively.

In light of the diminishing amount of upland nesting habitat within the agricultural landscape, and because the presence of adequate plant cover is crucial for successful nesting, hatching and brood rearing, it is expected that the greatest risk posed to ducks by herbicides is the reduction in the quality and quantity of cover. Thus, in this section, we attempt to quantify only the latter impact of herbicides on duck populations. In our calculations it is assumed that herbicides impact duck populations by reducing nest density and/or nest success.

The duck population under consideration is limited to the dabblers (84.4% of total; see Chapter 3) as they are exclusively upland nesters. The effects of herbicides on emergent wetland vegetation are generally not known so possible loss of nesting cover for diving ducks cannot be adequately estimated here. It was assumed that the sex ratio of dabblers is 1:1 and that all pairs attempt to nest. The total number of ducks under consideration is therefore 4.38 million pairs. The distribution of the nesting dabblers by census district, crop type and habitat has already been described in Chapter 3 (see Figure 3.5 and Appendix 3.9). The calculation of the number of successful nests produced on prairie farmland involves some modification and extension of the calculations carried out in Chapter 3. The parameters used in the present scenario are found in Table 8.1. The nest success rates are averages of published estimates (Table 8.2). The proportion of area sprayed for each crop type was calculated using values found in Table 4.13 and assuming that the use of grass herbicides was independent of the use of broadleaf herbicides.

The steps involved in the calculation of the total number of successful nests produced in each census district are as follows:

(1) the total agricultural area of the grassland-parkland region of the prairies is divided into the eight major land use types;

(2) the surface area of each major land use type is divided into its sprayed and non-sprayed portions (see Table 8.1);

Table 8.1. Parameter values entered in the calculation of successful nest production in the agricultural areas of the grassland-parkland region of the prairies.

Land Use Type	Total Area (x 000 ha)	Proportion of Area Sprayed	Habitat Type	Nest Density (no./ha)	Nest Success (%)
Wheat	11,619	91.5%	Crop	0.03	10
			Margin	0.3	40
Oats	1,022	65.0%	Crop	0.03	10
			Margin	0.3	40
Barley	4,209	86.1%	Crop	0.03	10
			Margin	0.3	40
Rye	389	20.0%	Crop	0.03	10
			Margin	0.3	40
Flax	448	90.8%	Crop	0.03	10
			Margin	0.3	40
Canola	1,141	92.3%	Crop	0.03	10
			Margin	0.3	40
Tame Hay	1,917	11.5%	Crop	0.3	40
			Margin	0.3	40
Fallow	9,044	0.0%	Crop	0.01	10
			Margin	0.3	40

Table 8.2. Nest success of ducks by habitat type.

Habitat Type		% Nest Success	Source
Grain crops and stubble	wheat crop	0	Evans & Wolfe 1967
	wheat stubble	0	
	grain stubble	0	Duebbert & Kantrud 1974
	mulched stubble	9	
	standing stubble	15	
	growing grain	40	
		mean = 10.7	
Forage	pasture and hay	25	Evans & Wolfe 1967
	alfalfa	21	
	hay	50	
	tame hay	64	Duebbert & Kantrud 1974
		mean = 40	
Fallow	summer fallow	8	Higgins 1977
Margins (undisturbed land)	roadsides	46	Evans & Wolfe 1967
	native grasslands	28	Kirsh 1969
	undisturbed grass-legume cover	68	Duebbert & Kantrud 1974
	idle grassland	25	Higgins 1977
	idle pasture	50	
	roadsides	19	
	idle alfalfa	22	
	dry wetland basins	44	
		mean = 37.7	

(3) the area of each sprayed and non-sprayed portion is divided into two habitat types: the crop (consisting of 90% of each portion) and the margin (consisting of 10% of each portion);

(4) to determine the number of nests initiated by the ducks in each of the 32 distinct portions, each area is weighted for its attractiveness to nesting hens by multiplying the area by the appropriate nest density (see Table 8.1); subsequently, each weighted area is divided by the sum of all weighted areas to obtain the proportion of the duck population nesting in each area. The number of pairs of ducks thus obtained is equal to the number of nests initiated in each area;

(5) again, for each of the 32 habitats the number of nests initiated is multiplied by the nest success to obtain the number of successful nests;

(6) the sum of the number of successful nests is calculated across all 32 habitats to obtain the total number of successful nests produced on agricultural lands for each of the census districts of the grassland-parkland region of the prairies.

The calculations above were first done, assuming no herbicide contamination of the margins, to determine "baseline" nest production. These results were compared to successful nest production determined under various degrees of reductions (1, 25, 50, 75 and 100%) in nest density and/or nest success in treated margins. The number of successful nests produced was therefore calculated for 25 combinations of reductions in nest density and nest success. Reducing nest density in sprayed margins allows for the redistribution of nesting ducks among the various habitatas before nest production is calculated.

While an accurate estimate of the amount of margin receiving herbicide drift is not available without adequate field studies of contamination, it was assumed that all margins in the treated areas will be affected in the long run because of the yearly application of herbicides. It was also assumed that margins receiving drift are exposed to the herbicide at the same rate as the crop. This is supported by Maybank and co-workers (1978b) who concluded that "... due to the additive nature of the droplet drift coming off successive swaths, ... the drift density at the downfield edge may well exceed the deposit density within the field ......" his is discussed in greater detail in Chapter 5.

In the scenarios, a number of simplistic assumptions were also made concerning duck breeding. For instance, the nest success rates used in the calculations represent the success of all nesting attempts, and therefore represent hen success or the probability that a hen will have one successful nest in one or more attempts (Cowardin et al. 1985). For the present exercise, the average number of nests produced per hen is set at 1.0. In reality, this number could be quite different. In their North Dakota study, Cowardin and co-workers (op. cit.) found that hens in their first year of nesting produced an average of 0.92 nests while hens in their second or later nesting years produced an average of 2.26 nests. Ducks may choose to nest in other areas of the prairies because of poor overall habitat conditions in agricultural areas or they may not nest at all.

The number of fledged young produced was not estimated in our scenarios because the fate of ducklings from hatch to fledging under various habitat conditions is not adequately understood.

It is important to note that the calculations described above clearly do not reflect actual duck production in the grassland-parkland region of the prairies. This exercise does not attempt to model duck recruitment during the breeding season as this would require a level of detail and sophistication inappropriate for the size of the area which is examined here. Instead, the focus is on gross estimates of nest production by land use and habitat type across the prairies, against which the effect of the deterioration of certain habitats through herbicide use can be measured. Thus, only the relative amounts of production are of interest here and not the absolute values. This approach is dictated in part by the spatial scale at which the herbicide use information is collected. Finer resolution of the use patterns within agricultural areas where duck production has been investigated would provide the basis for a more refined model.

8.2.2 Hazard scenarios - output

8.2.2.1 No herbicide contamination of margins or "baseline" scenario

In this scenario it is assumed that all herbicide applications take place without any significant contamination of non-crop areas.

The baseline production of successful nests (i.e. no reductions in nest success or densities) is described in Table 8.3. Examination of production by habitat type shows that the margins contribute 60.4% of all successful nests in spite of making up an estimated 10% of the total surface area. This is easily accounted for by nest densities and success rates which are, respectively, three and four times greater than in the crop itself. Crop areas contribute 39.6% of all successful nests primarily due to tame hay where nest densities and success rates are equal to those of the margins; if tame hay production is ignored, total crops contribute just under 10% of all successful nests.

Nest production by land use type is proportional to the surface area of that type except for tame hay (see Table 8.3). The latter contributes the most to total production, due to greater nest production in the crop, in spite of having a surface area less than a fifth of the area planted for wheat.

8.2.2.2 Herbicide contamination of margins

In these scenarios, it is assumed that the effect of herbicide contamination of margins rests on two variables: nest density (ND) and nest success (NS). These variables affect nest production in different ways: a reduction in nest density forces part of the hen population to nest elsewhere; a reduction in nest success within any one habitat decreases the number of successful nests produced for the hen population nesting in that habitat.

Table 8.3. Number of successful nests produced, by land use and habitat type, in the grassland-parkland region under a "no herbicide effect" scenario.

Land Use Type	Area (x 000 ha)	Number of Successful Nests[a] Habitat Crop	 Margin	 Total
Wheat	11619	97677 (7.0)	337003 (24.1)	434680 (31.1)
Oats	1022	6874 (0.5)	27831 (2.0)	34705 (2.5)
Barley	4209	22760 (1.6)	97276 (7.0)	120036 (8.6)
Rye	390	2565 (0.2)	10090 (0.7)	12655 (0.9)
Flax	449	2085 (0.1)	9101 (0.6)	11186 (0.7)
Canola	1141	7150 (0.5)	31780 (2.3)	38930 (2.8)
Tame Hay	1972	398755 (28.6)	44306 (3.2)	443061 (31.8)
Fallow	9044	27385 (2.0)	272753 (19.5)	300138 (21.5)
Total	29846	565251 (40.5)	830140 (59.5)	1395391 (100.0)

[a] Numbers in parentheses indicate the percent contribution to total successful nest production

Figure 8.2 illustrates the interaction between these two variables. If we focus on the effect of reducing only NS, we find that there is a linear decrease in the number of successful nests produced until the successful nest production within sprayed margins is down to zero (NS = 0) irrespective of the number of birds nesting within. The impact of reducing NS is more severe at higher values of ND. As illustrated in Figure 8.2, the lower the nest density is, the shallower the slope of the line relating successful nest production to nest success will be. This simply reflects the fact that, as the margin habitat becomes less attractive to nesting hens, fewer of them will nest within the habitat, thereby providing fewer nests from which the number of successful nests is determined. More nesting birds are forced into the fields proper where nest success is generally poor.

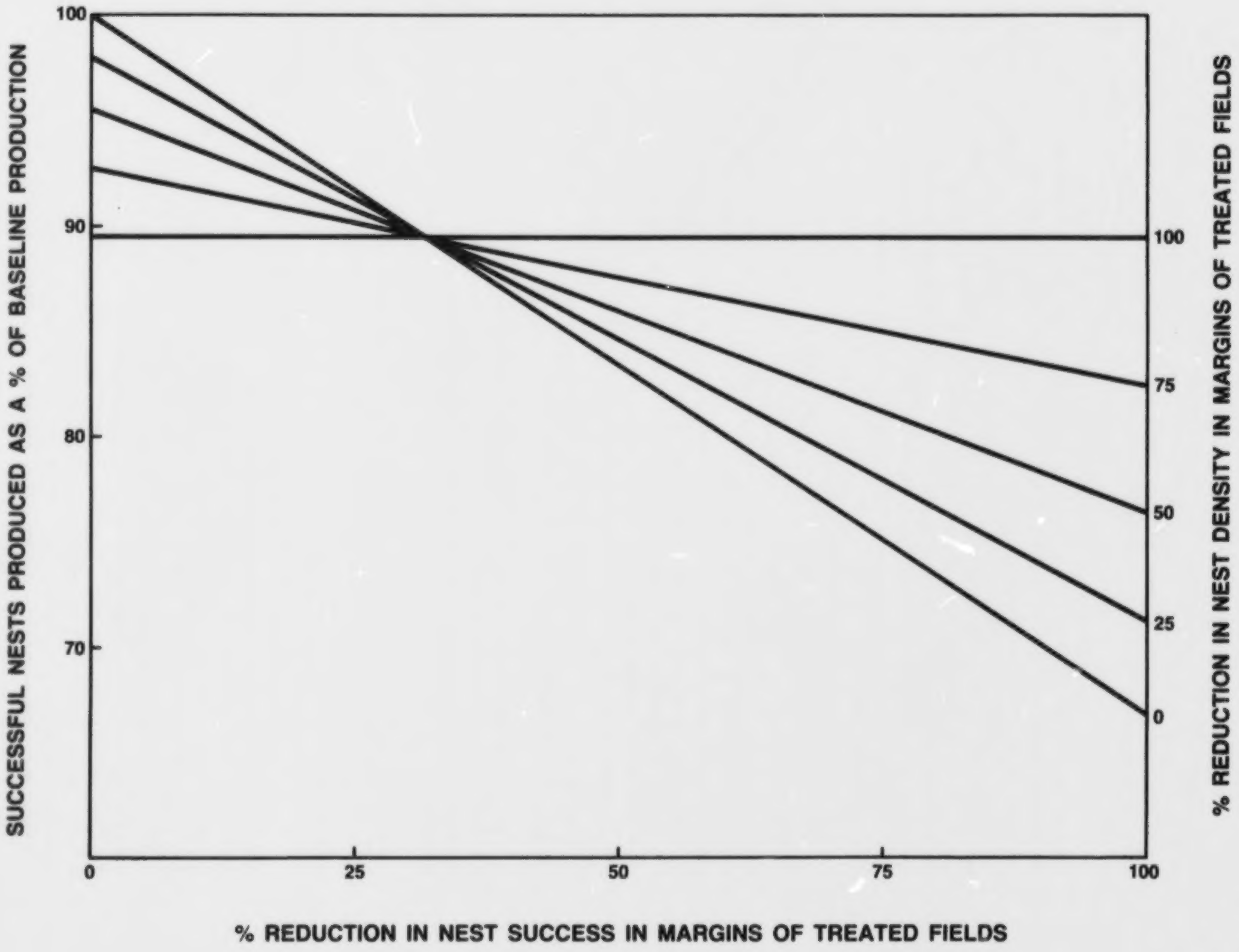
Given the proportions of areas sprayed presented in Table 8.1, we find a maximum potential loss of total successful nest production of approximately 33% under the scenario developed here (see Figure 8.2: % Δ ND = 0, % Δ NS = -100). However, it is not likely that nest success rates in the margins will fall below those in crops and that nest density will stay unchanged after herbicide contamination of the margins. Examination of the results from the study of Dwernychuk and Boag (1973) on the effect of 2,4-D application on nesting ducks and their habitat shows that nest densities fell by 75% one year after spraying. The authors related this decline to the loss of broad-leaved plants. If we apply this reduction in nest density within sprayed margins to the present scenario, we find that the decrease in successful nest production will range from 7.3%, when nest success is unchanged, to 17.6% when nest success is reduced to zero.

8.2.3 Discussion

One of the major points emerging from this analysis is the importance of margins in providing more than half of all nests produced in spite of representing only an approximate 10% of the land area. This is not surprising in light of the generally poor habitat available to ducks in agricultural areas. In a study of Mallard recruitment in an agricultural environment in North Dakota, Cowardin and co-workers (1985) found that hens rejected cropland for nesting in favor of "margins". These authors also found that success was so low that the population was not maintaining itself and was being supplemented by pioneering birds.

The assessment carried out in this section leads us to believe that herbicide use in the prairies has the potential to reduce successful nest production in agricultural areas of the Canadian prairie pothole region by as much as 18%. This impact is substantial in light of the present make-up of the agricultural landscape. As successful duck production becomes more and more limited to "marginal" areas in the prairies, herbicide contamination of these habitats may become an increasingly important factor in the decline of prairie duck populations. It is clear that the concern posed by herbicide use cannot be separated from the larger problems of habitat loss and deterioration.

Figure 8.2. Reductions in successful nest production of ducks under various scenarios of herbicide effects.


SUCCESSFUL NESTS PRODUCED AS A % OF BASELINE PRODUCTION
100
90
80
70
0
25
50
75
100
% REDUCTION IN NEST SUCCESS IN MARGINS OF TREATED FIELDS
100
75
50
25
0
% REDUCTION IN NEST DENSITY IN MARGINS OF TREATED FIELDS

The estimates of herbicide impact presented here may be conservative as the analysis was not extended to include brood success. Reductions in vegetative cover could also lead to decreased survival of ducklings between the moment they hatch and the time they fledge, due to increased hunting success by predators.

This assessment leaves some questions unanswered; the lack of detail in the analysis does not allow the identification of those species which are most threatened by the degradation of habitat. It is fair to assume, however, that species which require dense tall vegetation for nesting, such as the Mallard, the Teals and the Gadwall (see Section 2.5.2), would be more severely affected by loss of vegetative cover. Also, divers which nest in wetlands and along their borders can be expected to be less productive if wetland vegetation is affected by herbicide contamination. This latter concern could not be addressed within this assessment because of the paucity of information on the effect of herbicide drift on wetland plants (see Chapter 7).

Finally, it is important to note that certain aspects of duck behaviour were ignored in the development of the scenario. The actual nest densities obtained under some scenarios for the various habitat types were two to three times those used initially to apportion the duck population into each habitat. While the values obtained were two to three orders of magnitude lower than those of duck populations nesting successfully on islands (see Lokemoen et al. 1984), they are higher than nest densities recorded in agricultural environments. This indicates that, under this assessment, the birds were "crowded" into poor habitat where in reality they might have opted to avoid nesting or to move on to non-agricultural areas of the prairies. Under these circumstances, it would be very difficult to assess the extent of the impact of herbicide contamination of margins on duck production.

However simple, this assessment points to possible harmful effects of herbicide use on duck production in the Canadian prairies. This is of special concern in light of the loss and deterioration of prime nesting habitat.

8.3 Insecticides

8.3.1 Hazard scenarios - input

As in the case of herbicides, insecticides can have both a direct and indirect impact on duck populations. Earlier sections of this report touched very briefly on the acute toxicity of some major prairie insecticides, such as carbofuran which has directly caused waterfowl mortality when used on forage crops (Appendix 1.1). In 1985, carbofuran was used on an estimated 44% of all areas sprayed for grasshopper control. Spraying for grasshopper control is especially heavy along roadsides, fencelines and dry slough borders, all of which are favored nesting sites for upland-nesting ducks. If spraying and hatching coincide, mortality of the young brood will probably occur. Although carbofuran is not well adsorbed dermally, exposure is likely to be through a combination of dermal contact and ingestion of contaminated grasses and insects, including the grasshoppers themselves. Although the toxicity of carbofuran to waterfowl is approximately tenfold that of the next most toxic insecticide, methamidophos, demeton, trichlorfon, methomyl and methidathion should also be examined for possible toxic effects on ducks. The use of these insecticides, however, is usually small relative to the use of carbofuran, which is registered for use on most prairie insect pests and is virtually the only insecticide used on such pests as flea-beetle in canola. Carbofuran, however, is not registered for control of oil-seed pests such as bertha armyworm, cutworm and diamond-back moth. The 1971-72 bertha armyworm outbreak was handled almost exclusively with methomyl.

As seen in Chapter 6, several insecticides in current use have the capacity, when allowed entry into sloughs, to devastate the invertebrate community to such a degree (80% or more of the biomass) as to render the slough incapable of supporting duckling growth for that season. The most hazardous compounds, in terms of indirectly impacting the duck population in a "typical" slough of moderately alkaline nature, are the synthetic pyrethroids cypermethrin, deltamethrin and permethrin, as well as the organophosphates azinphos-methyl and chlorpyrifos. Moderately hazardous compounds include carbaryl, methoxychlor and, under some conditions (e.g. higher acidity), carbofuran and malathion. The most "ecologically" hazardous insecticides are some of the safest products from the point of view of acute toxicity to vertebrates (Appendix 1.1). This problem leaves regulators concerned with protecting the environment with difficult decisions.

The question being addressed in this section is how many ducks (dabblers and divers), because of their association with agricultural land, are likely to be at risk due to insecticide use. To answer this question, we have devised a series of scenarios portraying "average" insecticide use as well as intense insecticide use which may occur during severe pest outbreak situations. The latter consists of two scenarios, a major outbreak of grasshoppers and an outbreak of a major canola pest. These scenarios are described in Table 8.4. The resulting outputs are given in Table 8.5. These outputs are described in more detail below.

As seen above, dabblers nesting within crops proper (with the exception of those in forage crops) have a lower nesting success than those birds nesting in margin habitats, regardless of pesticide use. The chance of successful nesting in summer fallow is almost nil, although some birds undoubtedly try. In a number of our scenarios, especially those dealing with indirect impacts which affect nesting success rather than adult survival, it would be "unfair" to consider a duck nest at risk from pesticide use when its chances of success are limited even in the absence of any such use. In the case of dabblers, the number of successful nests is obtained by correcting nest density estimates by success rates based on literature values (Table 8.2). As with the herbicide scenarios, initial nest density for dabblers is calculated from Appendix 3.9, assuming that the sex ratio is 1:1 and that all birds attempt breeding in any given year. In the case of divers, it is assumed that the probability of nest success is independent of crop type (a reasonable assumption if most divers nest in association with marsh vegetation within wetland margins); thus the values presented in Appendix 3.9 are used directly without correcting for nest success.

8.3.2 Hazard scenarios - output

8.3.2.1 Annual insecticide use - non-outbreak situation

There is really no such thing as an average year for insecticide spray programs in the prairies. However, assuming that the year 1980 was a year more representative of average insecticide use than one characterized by major pest outbreaks (Section 4.2.2), we were able to calculate the number of ducks at risk with respect to the region-specific use of pesticides for that year based on the hazard scenario (outlined in Table 8.4) and the population estimates (Appendix 3.9, Figure 3.5). As a rough estimate, 0.32 million dabblers (approximately 2.3% of the estimated 14.1 million Canadian prairie dabbler population) nesting in agricultural crops proper were sprayed directly with insecticides; a further 0.13 million nesting in close proximity to field crops were exposed to drift. Drift was especially likely in 13% of cases, the estimated proportion of insecticides applied aerially. The direct consequences to ducks of being in the spray area are difficult to ascertain, especially in the absence of firm data on the insecticides used. Adult mortality has been recorded following sprays of carbofuran. Direct toxic impacts are actually more likely on young ducklings due to their small size and higher surface to volume ratio. We estimate that under our scenario conditions (Table 8.4), in an average year for nesting success, 50,000 broods are directly within the spray area or exposed to the spray drift. This corresponds to 3.6% of the total dabbler production associated with cropland. (Our model assumes that, in an average year, 1.4 million broods are produced from the 8.5 million dabblers nesting in association with cropland. This figure is derived from the estimates in Appendix 3.9 corrected for nest success data outlined in Table 8.2.). The actual impact on the broods will depend on the extent to which the broods are exposed to the insecticide spray and this, in turn, will depend on the degree of temporal overlap between brood movements and spray programs.

Adult divers and their broods will similarly be directly exposed to insecticide sprays, especially those applied late in the season (i.e. July and August). Under our scenario conditions, we estimate that 45,000 divers (or 3.4% of the diver population in cropland) are nesting on wetlands within the area sprayed with insecticides. Again, direct impacts were especially likely where aerial application was involved (an estimated 13%).

Both dabblers and divers are further at risk with respect to the effects of insecticides on their food supply. In the case of dabblers, 13% of the 50,000 broods calculated above are expected to have their feeding habitat directly oversprayed with insecticides. Thirteen percent of the broods of the 45,000 adult divers calculated above are expected to be similarly impacted. This corresponds to about one-half of one percent each for the dabbler and diver production associated with cropland. Impacts are likely to occur when insecticides of high invertebrate toxicity such as the synthetic pyrethroids, and the organophosphates chlorpyrifos and azinphos-methyl are used. Although our estimate of the number of broods at risk is low, it will increase proportionately with rise in the use of aerial application. Furthermore, an unknown but potentially substantial proportion of the remaining broods will be impacted through drift resulting from ground application of the insecticides in close proximity to the sloughs

Table 8.4. Hazard scenarios with insecticide use.

Conditions of scenario	Annual use - non-outbreak situations	Major grasshopper outbreak	Major outbreak of pest of canola (bertha armyworm or diamond-back moth)
Type of area treated	- cropland only	- all lands, both crop and non-crop (Section 4.2.1)	- canola crop only
Location of insecticide use	- taken from Statistics Canada (1981) which gives insecticide use by census division for year 1980 (Section 4.2.2) - impossible to determine which types of crops are more likely to be sprayed on account of spatial and temporal variability of insect problems, although it is assumed that no spraying takes place on fallow	- infestation confined to the following waterfowl strata: 26 and 27 in Alberta; 30, 32 and 33 in Saskatchewan; and 38 in Manitoba (Figure 3.1) - based on annual reporting in the "Canadian Agricultural Pest Review" (4.1.2)	- infestation confined to the following census districts: 7 and 10 in Alberta; 11 to 17 in Saskatchewan; 16, 17 and 20 in Manitoba - area based on reports in the "Canadian Agricultural Pest Review" concerning the 1971-72 bertha armyworm outbreak
Surface area treated	- total of 1.0 million ha sprayed (reapplication frequency unknown) as follows: Saskatchewan - 0.61 million ha Alberta - 0.17 million ha Manitoba - 0.24 million ha	- the year 1985 taken as worst-case scenario (Table 4.5): Saskatchewan - 2.8 million ha Alberta - .53 million ha Manitoba - .30 million ha	- worst-case estimates from 1971-72 outbreak of bertha armyworm and and 1985 outbreak of diamond-back moth (Section 4.2.2): Saskatchewan - .45 million ha Alberta - .09 million ha Manitoba - .02 million ha
Time of year and overlap with waterfowl breeding	- all summer long - overlaps with entire breeding season	- mid-June-August - overlaps with most of breeding season	- mid-July-August - overlaps with brood rearing period of re-nesters and primarily with diving ducks (Figure 2.8)

Table 8.4 (cont'd)

Conditions of scenario	Annual use - non-outbreak situations	Major grasshopper outbreak	Major outbreak of pest of canola (bertha armyworm or diamond-back moth)
Method of application	- 13% by aerial application (Section 5.1)	- 13% by aerial application (Table 4.5)	- 100% by aerial application
Likely insecticides of choice (based on recent use and predictions) (Chapter 4)	- mixture	- carbofuran - deltamethrin - carbaryl - chlorpyrifos	- deltamethrin - methomyl - chlorpyrifos - malathion
Conditions under which the number of breeding ducks at risk is calculated	Direct impact - main impact through insecticide deposit in nesting habitat and subsequent exposure of adult and especially young - number of ducks within spray zone estimated from the proportion of sprayed to unsprayed cropland in each census division - ducks nesting in crop proper most at risk (e.g. dabbler numbers in Appendix 3.9) - other ducks at risk from drift only. This drift likely to be considerable at least for 13% of application which takes place by air - choice of insecticide difficult to predict	Direct impact - spray areas thought to coincide with areas heavily utilized for nesting margin. Assume that all margin areas in spray zone likely to receive direct spray. Reapplication to margins increases likelihood that hatch and spray will coincide - conservative estimation if assumed that number of ducks within spray zone proportional to fraction of stratum that is sprayed - at least half of spraying with very toxic compound (e.g. carbofuran) - number of ducks assumed to be average relative to other years (Section 3.4.3)	Direct impact - ducks nesting within crop sprayed directly - high probability of drift in margins since all applications aerial - number of ducks within spray zone estimated from the proportion of sprayed to unsprayed canola - possibility that most of the spray will be insecticide of high acute toxitity (e.g. methomyl)

Table 8.4 (cont'd)

Conditions of scenario	Annual use - non-outbreak situations	Major grasshopper outbreak	Major outbreak of pest of canola (bertha armyworm or diamond-back moth)
Conditions under which the number of breeding ducks at risk is calculated (cont'd)	Indirect impact - main impact through contamination of sloughs - in all cases of aerial application (13%), direct contamination of wetlands very likely - other wetlands subject to drift - number of ducks whose habitat (i.e. sloughs) are at risk calculated from the proportion of sprayed to unsprayed cropland as above	Indirect impact - any sloughs present have 13% chance of overspray - drift possible in other cases - at least 25% of spray has potential for severe aquatic impact - water can be present in waterfowl stratum when spraying taking place but exact distribution of water with respect to spray area is uncertain	Indirect impact - all sloughs in cropland subject to overspray - major impact on diving duck popuation because of late spray season and higher dependence of divers on invertebrate food supply - toxicity of one of the major insecticides (methomyl) poorly characterized. All other candidates have high likelihood of aquatic impact

see Chapter 5). It is also possible that indirect impacts will affect breeding females since they too are dependent on the invertebrate food supply for reproduction. An impact is likely when late season sprays or repeated sprays preclude a full recovery of invertebrates, and therefore, reduce the supply of invertebrates available for overwintering and spring recolonization of sloughs.

8.3.2.2 A major grasshopper outbreak

Because of both their extent and frequency of occurrence in the prairies, grasshopper sprays are potentially the most disruptive to nesting waterfowl. Since both crop and non-crop areas are commonly treated, we cannot exclude, as was done in the previous section, those ducks which nest in non-crop areas and assume that they are immune from any spray impact. Our grasshopper scenario is, therefore, quite crude in comparison to the non-outbreak scenario and we believe that it is also very conservative, at least as far as the direct toxic impacts on upland-nesting dabblers are concerned. The reason for this is that spraying tends to be concentrated in field or slough margins and roadsides, all of which are favoured nesting areas. Our scenario, on the other hand, assumes that the likelihood of spraying of margins is only one-ninth as likely as field spraying, given our assumed 9:1 ratio in crop to margin surface area. In the absence of any data indicating how the total grasshopper spray is divided between crop, margin and non-crop area, we cannot safely assume otherwise.

One of the most critical questions in this scenario is whether there are any ducks nesting in the area of infestation in those years when grasshoppers are in an outbreak condition. A common view expressed to us during the drafting of this report was that grasshopper outbreaks are associated with drought conditions, and thus an overlap between spraying and nesting ducks was unlikely. The information presented in Section 3.4.3 and 4.2.1 indicates that duck numbers have, in fact, been average during the large grasshopper spray programs of 1974 and 1982 in southern Saskatchewan. However, we must acknowledge that the air-ground waterfowl transects on which these duck data were based, may not be representative of the habitat, as discussed in Section 3.2.4. In the absence of any better information, we maintain that overlap between grasshopper spraying and waterfowl nesting is likely because of the large geographical areas involved during grasshopper outbreaks. We also maintain that once an outbreak is underway, spraying can be positively correlated with summer rains since, under rainy conditions, farmers have a fair expectation of return from any costs accrued through spraying (Section 4.2.1). The rains will also have the effect of preserving more semi-permanent sloughs and hence more habitat will be available for nesting and brood rearing. It is imperative that the relationship between grasshopper spraying and duck nesting be further investigated. Only then will we be confident in our risk estimates.

Given the scenario outlined in Table 8.4, where the infestation is limited to strata 26, 27, 30, 32, 33 and 38, we estimate that 16%, 5.3% and 20% of the infested land area will be sprayed in Saskatchewan, Alberta and Manitoba, respectively. We estimate that 0.93 million dabblers and 0.13 million divers, or 6.9% and 6.2% of their respective Canadian prairie populations, will be within the spray areas. (Note: These percentages are based on Appendix 3.1 totals which differ slightly from those given in Table 8.4. Comparison between this scenario and the other two should be made on the basis of proportions rather than absolute duck numbers.) As mentioned above, the overlap between upland nesting dabblers and spray is thought to be extensive. In 1985, half of the total spray was carbofuran which represented a direct acute hazard for 0.46 million nesting dabblers and however many broods they managed to produce. Diving ducks were probably only at direct risk with respect to aerial application (thought to be about 13% of all areas treated).

Indirect impacts via the invertebrate food supply of ducklings were likely at least where deltamethrin was used, which represented 22% of the total area treated with insecticides applied in 1985 (Section 4.2.1) and probably where aerial application was used and resulted in direct overspray of the habitat. As a result, we estimate that, at the very least, 0.19% (0.13 x 0.22 x 0.069) of the prairie dabbler habitat (27,000 breeding birds) and 0.18% (0.13 x 0.22 x 0.062) of the prairie diver habitat (3,700 breeding birds) received direct applications of deltamethrin. Given the information presented in Chapter 6, it is probable that this habitat was degraded to the extent that it was not able to support any growth of broods. These numbers are low in comparison to the number of ducks in the spray area but they will increase if: (1) aerial application becomes more popular; (2) deltamethrin and other insecticides of high invertebrate toxicity (e.g. chlorpyrifos) become more commonly used; and (3) synthetic pyrethroids and other compounds of high invertebrate toxicity but low vertebrate toxicity become the material of choice for aerial delivery.

Indirect impacts resulting from ground application are difficult to predict but may be high on account of drift (Chapter 5).

8.3.2.3 An outbreak of a major canola pest

Given a scenario of a late (mid July-August) spray program for bertha armyworm or diamond-back moth, the species most at risk are the diving ducks nesting in association with wetlands in the canola crop. On the basis of the input parameters given in Table 8.4 we estimate that 43,000 divers, or approximately 2% of the Canadian prairie population, will be at risk from such a spray program. Direct overspray is almost assured given 100% aerial application. If direct impacts occur, they are likely to be found in those areas sprayed with methomyl which is the most acutely toxic of the insecticides used. Indirect impacts are most probable with deltamethrin and chlorpyrifos and likely with malathion and methomyl, although inadequate data for the latter prevented its inclusion in the model in Chapter 6. Given the prolonged dependency of divers on an adequate invertebrate supply, the exact temporal overlap between hatching and spray is less critical than in other scenarios. Insecticide sprays are likely to impact at any time during the brood rearing period. A possible mitigating factor to the above scenario is that diving ducks tend to favor larger wetlands for nesting (Section 2.5.2.2), where the likelihood of direct overspray may be lower.

In years when dabbler nesting is delayed, they too may be affected by this late spray program. Re-nesting in spray areas is also likely to be hampered. An estimated 174,000 dabblers nest within the spray area (1.2% of the Canadian prairie population). These birds are expected, in an average year, to produce 22,500 broods or 1.6% of the total dabbler brood production and it is that production which would be at risk through an aerial overspray of the habitat.

8.3.3 Discussion

In terms of magnitude, grasshopper control programs have the potential for the greatest impact on the waterfowl resource (Table 8.5). Unfortunately, we are the least confident about the true impact of these programs given the large number of uncertainties involved. Although the percentage of ducks at risk from pesticide use is generally low (usually 2-4%) relative to the yearly fluctuation in duck numbers, it is important to realize that the above scenarios are not mutually exclusive; they are additive. All three scenarios occurred in 1985 which means that, if our estimates are correct, approximately 9-10% of the breeding prairie dabbler population nested within the actual insecticide spray zone. Furthermore, any impacts resulting from insecticide use are additive to impacts stemming from other agricultural disturbances. As the availability of nesting and feeding habitat continues to shrink, the quality of the remaining habitat will assume greater importance. For example, in two of the three scenarios above, we have assumed that 10% of the total land area within the agricultural cropland was available as high quality refuges to nesting ducks. It is quite probable that this is a gross overestimate for many parts of the prairies where fence lines and hedgerows have been removed and where crops are sown right up to the grades of roads and highways. Any reduction in the available upland refuges will result in a proportionate increase in overlap between nesting and cropping which will have the effect of increasing the number of ducks at risk from insecticide use.

8.4 Conclusion

As seen in this chapter, the potential impacts of pesticide use in the Canadian prairies are not necessarily inconsequential. Using all the information available, this paper exercise showed that contamination of field margins and other non-crop habitat within the agricultural landscape with herbicides can result in a loss of nests to dabbling ducks of 7 to 18% of the total production associated with croplands. This represents 98,000 to 246,000 fewer broods than can potentially be produced successfully in these areas. Similarly, we further estimated that in one summer, albeit one which will be remembered by farmers for its pest outbreaks, approximately 9-10% of all nesting dabblers were directly within the insecticide spray zone. Exact losses to the insecticides are difficult to predict but the toxicity of the most popular product to ducks is such as to be a real cause for concern.

Table 8.5. Summary of ducks affected under three scenarios of insecticide use.

	Scenario of Insecticide Use		
	Annual Use	Grasshopper Outbreak	Canola Pest Outbreak
Dabblers			
No. of birds sprayed directly	320,000	930,000	174,000
% of Canadian prairie population	2.3%	6.6%	1.2%
No. of broods sprayed	50,000	142,235	22,500
Divers			
No. of birds sprayed directly	45,000	130,000	43,000
% of Canadian prairie population	2.0%	6.2%	2.0%
Total			
No. of birds sprayed directly	495,000	1,060,000	217,000
% of Canadian prairie population	3.0%	6.3%	1.3%

Managers and regulators face a very real problem in coming to grips with environmental factors as elusive as pesticides. The impact of herbicides, for instance, need not be sudden and dramatic but rather is expected to build up over the course of several years of repeated impact on non-crop areas. In addition, the scenarios developed here estimate decreases in nests successfully hatched which is very difficult to verify in the field or through analysis of the population dynamics. Even instances of direct mortality resulting from the use of acutely toxic insecticides are unlikely to come to light without a great deal of specialized research. As documented above, the impact of any one spray program is usually small (i.e. in the 2-4% range) relative to yearly fluctuations in the population estimates. Furthermore, this impact is spread out over thousands of hectares and by and large goes unnoticed.

One can get a better sense of the effect of pesticide use on prairie waterfowl if these numbers are compared to harvest levels, a much better studied source of mortality. It is estimated, based on the National Harvest Survey carried out by the Canadian Wildlife Service (Métras 1986), that the retrieved kill of 11 species of ducks studied in this report averaged just under 1 million birds per year between 1982 and 1985 in the three prairie provinces. Therefore the number of these ducks harvested in the prairies by hunters represents approximately 6% of the Canadian prairie population which is well within the range of the proportion of prairie birds sprayed with insecticides (Table 8.4). It is thus clear that the effect of pesticide use on prairie duck populations may be on the same scale as the effect of Canadian prairie hunters on these same populations. In the present context of reduced habitat quantity and quality, low population levels and high harvest rates, pesticides may be having a greater and more subtle impact than can be simply estimated here.

CHAPTER 9

SUMMARY, RECOMMENDATIONS AND RESEARCH NEEDS

The pothole region of the western prairies of Canada is both intensively cultivated agricultural land and productive waterfowl habitat for the 16 million ducks which breed there annually. As agriculture has expanded and intensified in the past 100 years, prime waterfowl habitat has declined both in quantity and quality. Concurrently, pesticide use has increased tremendously since the late 1940's. In the context of diminishing natural habitat and declining duck populations, the impact of the agricultural use of pesticides on the reproductive success of prairie ducks is of concern.

As newer pesticides with lower direct toxicity to vertebrates are developed, concern over direct effects on birds should diminish. At the same time, these newer products are more toxic to target and related organisms and, therefore, pose an indirect threat to waterfowl through reductions in food or vegetative cover.

It was the goal of this report to assess the impact of pesticide contamination of waterfowl habitat on duck reproductive success. After reviewing the basic ecology of ducks and their habitat in the Canadian prairies, we attempted to estimate the number of ducks and the amount of duck habitat exposed to pesticides and thus, what effect pesticides might be having on duck productivity. In order to carry out the hazard assessment, the use, environmental behaviour and toxic impact to invertebrates and plants of a number of major use pesticides were reviewed.

9.1 WATERFOWL AND PRAIRIE SLOUGH ECOLOGY (CHAPTER 2)

9.1.1 Summary

There were three main objectives to Chapter 2. First, it was necessary to establish the physical and chemical nature of potholes along with their distribution in both space and time as these factors are important in assessing the likelihood of pesticide contamination and the fate, persistence and toxicity of these compounds. Second, the ecology of prairie sloughs was reviewed with particular emphasis on the processes which are critical to breeding waterfowl: the growth and production of invertebrates and vegetation in and around potholes. Finally, in the last section, we gave an overview of the ecology of waterfowl species breeding in the Canadian prairie pothole region; in particular the requirements of the breeding females and the ducklings for food and nesting cover were emphasized. This allowed us to identify the periods, from the time the eggs are laid to the moment the ducklings fledge, when the duck recruitment process is most sensitive to perturbations in the habitat.

The Canadian prairie pothole region covers over a half million square kilometers in the southern half of Alberta, Saskatchewan and Manitoba. The movement of the continental ice sheet during the last glaciation left behind a landscape characterized by poor drainage and millions of small depressions which collect water. The vegetation is typically grassland to the south and parkland further north. Pothole density varies from 4 to 40 per km^2, with an average of 12 per km^2, and is greater in the parklands than in the mixed prairie to the south. Approximately 85% of the pothole basins are less than 0.5 ha in size.

An arid climate combined with poor drainage leads to large fluctuations in slough numbers both within and between years. Factors such as snowmelt, rainfall, groundwater seepage, soil moisture, temperature and wind become critical in determining the water regime and, therefore, the permanency of ponds.

Except for pH which tends to be moderately alkaline (7.7 to 9.7), the concentrations of most chemical constituents tend to be highly variable among potholes. This is due to the heterogeneous nature of glacial deposits and also to the importance of groundwater in the water regime of many sloughs. Conductivity (an indicator of salinity) ranges from less than 50 to more than 20,000 μmho/cm. Pond bottoms tend to have a very high content of organic matter except in deeper sloughs where the sediments are finer and consist of loam, clay or sand.

The plant composition of prairie sloughs is intimately linked with the water regime of the pond. The dominant forms of vegetation are indicators of water depth and of the duration of flooding. Plants such as grasses, sedges, forbs and willows are characteristic of ephemeral ponds and of the outer edge of deeper ponds. Typical marsh species such as cattails, reeds and bullrushes are found in shallow and deep marshes of temporary and seasonal potholes and in the shallower areas of semi-permanent and permanent wetlands. Submergent species such as pondweed and water-milfoil are common in the deep open water zones of larger sloughs where flooding persists.

Water regime, salinity and disturbance (e.g. mowing, grazing, cultivating), in that order of importance, are the most important factors affecting the plant species composition of potholes. It is therefore not surprising, in light of the dynamic nature of water levels in prairie sloughs, that the vegetation of these ponds is continually changing from wet meadow to marsh to open water or vice versa, creating complex vegetation patterns which are highly variable between ponds.

Prairie sloughs have extremely high levels of primary production. Marsh habitats or wetlands are among the most productive ecosystems in the world: production values exceeding 2,000 g dry weight/m^2/year are not uncommon. These high levels of production originate from two major characteristics of sloughs which have very strong effects on the two essential nutrients, nitrogen and phosphorus. First, because of the lack of drainage, prairie sloughs are sinks which accumulate nutrients and organic matter from the watershed. This is why pond productivity usually reflects watershed fertility. Second, the alternation between flooded and dry conditions, along with the oscillation between aerobic and anaerobic conditions typical of shallow bodies of water, enhance the decomposition of organic matter and the subsequent release of nutrients from the bottom. This rapid recycling of nutrients favors rapid growth and high levels of production.

The combination of high levels of primary production, a rich flora, a structurally complex and variable vegetation along with fluctuating environmental conditions favors the development of a very rich and productive invertebrate fauna. The high levels of invertebrate abundance in prairie sloughs are a function of the large surface area provided by both the living and dead vegetation. Invertebrates associated with plants are much more numerous than those found on the bottom or in the water. The dominant taxa are gastropods, amphipods, notonectids, corixids, larval chironomids, trichopterans, odonates and ephemeroptera. The strong association between plants and invertebrates is reflected in the high variability of the standing crop of invertebrates between and within potholes. Changes in numbers seem to follow changes in the vegetation. Seasonal maxima of invertebrate numbers and biomass occur during the peak development of the vegetation in late May to late June.

While there is a large variety of feeding mechanisms used by invertebrates, the dominant food item is algae, either as phytoplankton or periphyton. Little vascular plant tissue is consumed. The dominant role played by invertebrates in the slough ecosystem is in the breakdown of the litter into smaller particles. This stimulates microbial decomposition of the plant litter. The size of sediment particles is also modified by the action of sediment feeders which causes a mixing of the sediments. All of these feeding activities lead to a high rate of nutrient recycling which is required to maintain the high levels of production characteristic of prairie sloughs.

Of the more than 20 duck species which are known to nest in the pothole region, only 11 have breeding populations in excess of 100,000 individuals. The review of waterfowl ecology focused on those 11 more common species which consist of seven dabbler species (Mallard, Pintail, Blue-winged Teal, Northern Shoveler, American Widgeon, Gadwall and Green-winged Teal) and four diver species (Lesser Scaup, Redhead, Canvasback and Ruddy Duck). Numerically, the dabblers are clearly the dominant group, comprising more than 80% of the prairie waterfowl population.

The Mallard and Pintail are the first to arrive on the breeding grounds in late March and the Ruddy Duck is the latest, arriving in late May or early June. Weather conditions usually determine when the birds first arrive. Mallards and Pintails are the first to initiate nesting, usually soon after their arrival. Lesser Scaup and Ruddy ducks do not initiate nesting until the latter half of June. If early attempts fail, most species will renest if conditions are good.

Clutches of early nesting species hatch between late May and mid-June and the ducklings usually fledge in late July to early August. Broods of diving species may still be flightless in October.

Breeding waterfowl have two basic requirements which are critical for successful recruitment: a complex of potholes to provide food throughout the breeding season and adequate vegetative nest cover adjacent to the sloughs. Species-specific habitat preferences are not very strong, although, as groups, the dabblers and divers differ considerably. Dabblers prefer shallow, temporary or seasonal wetlands while divers prefer deeper, larger and more permanent sloughs. In all cases, the presence of a well developed emergent aquatic flora is favored. Also, for dabblers at least, a complex of different wetlands may be essential to their success. The more shallow ephemeral ponds provide abundant food earlier in spring while deeper ponds, which thaw later in the season, supply invertebrates for the ducks when other ponds have dried up.

Ponds are a critical source of animal protein. Regardless of what foods are consumed during the rest of the year, during egg formation hens need a large component of invertebrate foods in their diet to meet their high protein requirements. The levels of protein and calcium in the diet during this time directly affect clutch size, hatchability and energy reserves available to newly hatched ducklings. The hens tend to select strongly for aquatic insects, crustaceans and snails. In the first few weeks after hatching, ducklings feed almost exclusively on aquatic invertebrates in order to maintain a high rate of growth which is essential to their survival. There is some evidence that food may be limiting in some ponds and that hens move their broods to other sloughs in search of sufficient food. Thus, a complex of wetlands may be important for successful waterfowl recruitment, as stated earlier.

In the Canadian prairies, agricultural cultivation has tended to confine ducks and their predators together in small areas of suitable habitat. Predation is the major source of egg mortality and nest loss. Abundant undisturbed nesting cover is, therefore, critical for the survival of incubating hens, eggs and broods. This requirement is reflected in their choice of nesting habitat. All dabblers and Lesser Scaup are primarily upland nesters while the other divers choose to nest over water among emergent vegetation. While nesting cover preferences are species-specific, undisturbed stands of grasses, buckbrush, snowberry, whitetop, clover, willow, sedges and aquatic emergents are frequently chosen by most species except for Pintails which will nest in low or sparse vegetation.

Little undisturbed upland habitat is left in the agricultural Landscape of the prairies. Also, fields are cultivated as close to the edge of the ponds as possible, leaving little margin in which birds can nest. Therefore, the success rates of waterfowl recruitment is intimately linked to agricultural activities. The habitats with the highest nest densities and nest success rates are idle grasslands, roadsides, untilled dry wetlands, ditches and hedgerows.

9.1.2 Recommendations and Research Needs

A lot of information is available on wetland ecosystems. Hydrologists have studied the water regime, chemists have looked at the water chemistry, botanists have studied the plants and waterfowl biologists have studied the foods of ducks. In spite of these efforts, when one attempts to relate the process of waterfowl production to the slough ecosystem, many questions seem to be unanswered, primarily because ducks are very mobile and make use of not just one pond but of a complex of various wetland types. It is important that studies focus on such a complex of wetlands to determine:

(1) the variability in primary production over time and between ponds;

(2) the variability in secondary production between ponds and its relation to the population dynamics of the dominant invertebrate species; this work should be related to (1);

(3) the capacity of sloughs to support ducklings and the effect of food abundance and its variability on duckling behaviour and survival.

9.2 WATERFOWL HABITAT DISTRIBUTION AND AGRICULTURE OVERLAP (CHAPTER 3)

9.2.1 Summary

Aerial waterfowl surveys, along with limited ground-truthing, have been conducted yearly since 1955. In this chapter, data averaged over the years 1970 to 1983 were presented. It is estimated that the Canadian prairies account for approximately ½ of the total North American breeding duck population. According to those same estimates, 84% of those ducks are dabbling ducks (tribe Anatini), especially Mallard, Pintail and Blue-Winged Teal. Breeding densities of Mallard, Gadwall, Widgeon, Shoveler, Green-winged Teal and Lesser Scaup are highest in the parklands, especially in Alberta. Densities of Pintail are highest in the grasslands of southern Alberta.

There is no comprehensive habitat inventory that includes small, temporary and semi-permanent basins of greatest importance to nesting waterfowl. The only prairie-wide inventory that is applicable to waterfowl is the Canada Land Inventory (CLI) begun in 1963. According to the rating scheme used, the best waterfowl lands (Classes 1-3) are situated within the aspen parkland zone along a broad diagonal strip running northwest to southeast from Edmonton to southwest Manitoba. A disproportionately large part of this prime habitat (53% of Class 1-2 and 64% of Class 3) lies in Saskatchewan.

The Canada Land Inventory for agricultural potential was also used to examine the distribution of agricultural lands in the Canadian prairies. According to the 1967 CLI data on "present land use", Alberta has the smallest proportion of the best agricultural lands (Class 1-3) in production. Saskatchewan shows the most extensive cultivation of marginal and very marginal agricultural lands. The trend to conversion of aspen parkland to agricultural land may be accelerating. If waterfowl and agriculture are at war then Saskatchewan is undoubtedly the main battle zone. It has the best and greatest waterfowl potential and yet the most intensive cultivation of all but the very poorest agricultural lands.

One of the goals of this chapter was to obtain a measure of overlap between waterfowl nesting and specific crop types or cultural practices. This was achieved by combining the aforementioned agriculture and waterfowl subsets of the CLI with agricultural statistics and actual waterfowl counts in the form of yearly survey data.

Following the unpublished work of Patterson, breeding duck densities were assigned to CLI Waterfowl Classes. Breeding duck densities ranged from 69-85 ducks/km^2 for Class 1-2 land, 36-62 ducks/km^2 for Class 3, 14-40 ducks/km^2 for Classes 4-5 and 6-9 ducks/km^2 for Classes 6-8. Although the best lands (Classes 1-3) account for ¼ of the total prairie land area, they comprise approximately ½ of all waterfowl nests. Saskatchewan accounts for ½ of the total waterfowl nesting, divided equally between Class 1-3 and Class 4-5 waterfowl lands.

On the basis of the agriculture and waterfowl CLI data, it is possible to calculate the overlap between the two resources. It was thus estimated that 63 and 80% of Waterfowl Class 1-5 lands overlap with Agriculture Classes 1-3 and 1-4, respectively. The impact of placing all Class 4 agriculture land into production would be greatest on the best waterfowl lands (Class 1), increasing the overlap of waterfowl land with agriculture by 21%. This analysis is tempered by the fact that much of the good waterfowl land had already been eliminated by agricultural activity before the CLI system came into effect. The proportion of the very best waterfowl habitat (Classes 1 and 2) overlapping with the best agricultural land (Classes 1-3) is highest in Manitoba (87%) intermediate in Saskatchewan (62-84%) and lowest in Alberta (57%). This east to west decline contrasts the decreasing trend in pond occupancy rate (the number of breeding ducks in May/the number of ponds in May) from west to east. It is tempting to speculate on a causal relationship.

Applying duck density estimates to these overlap figures, we estimated that 56% of all prairie breeding ducks nest in close association with "current" agriculture defined as Classes 1-3. The proportion would rise to 75% if all Agriculture Class 4 lands were placed in production. Taking this analysis one step further, we estimated the overlap between waterfowl and agricultural potential for each census district. We then estimated the overlap between waterfowl and a number of major crop types by "assigning" breeding ducks to those crop types based on probable nesting preferences and crop distribution information obtained in the quinquennial agricultural surveys. It is recognized that the majority of ducks do not nest within the cropland proper but rather in field margins, roadsides and slough edges. We have attempted to differentiate those birds nesting within the crop from those nesting on non-crop sites but nevertheless in close association with cropland. This close association is likely to be important if the aquatic sites chosen for brood rearing are surrounded by cropland. This very circuitous analysis was made in the absence of any attempts (published or otherwise available) to correlate directly the yearly waterfowl information with agricultural activities. Until such an analysis is undertaken, our conclusions must be tempered by the uncertainty of many of the assumptions underlying the calculations.

The results from this analysis indicate that, not surprisingly, wheat accounts for the largest number of ducks associated with cropland (35.5%). However, given a marked preference by nesting dabbling ducks for forage crops (25.1% of the nesting population), this crop type is second in importance after wheat. Densities of both dabbling and diving duck species are thought to be highest in association with forage crops, followed by canola. The latter crop is probably more important to diving ducks on account of its distribution within the parklands. The tabular information presented in Chapter 3 allows for an estimate of the possible impact of pesticides or any other agricultural disturbance on the waterfowl resource.

Since the early 1960's, the Canadian prairies have been undergoing a loss of wetlands similar to that which occurred in the U.S. at the beginning of the century. The trend shows no abatement and stems from the prevailing attitude of farmers and agriculturalists that wetlands offer no tangible benefit unless they can be drained and the land "improved" for agriculture. Only future study will resolve whether the practice of draining sloughs and eliminating margins has negative consequences to prairie agriculture. The disastrous effects on waterfowl are obvious.

It has been forecasted that grain production in Canada will increase from 35.6 million tonnes in 1981 to 50 million tonnes by the year 1990. An estimated 21% of this increase will come from putting new (mostly marginal) agricultural lands into production, thus signalling more losses to waterfowl. Although tangential to this report, habitat loss is an important factor in our assessment. As the habitat base for waterfowl is reduced, the quality of the remaining habitat becomes all the more important. Pesticides certainly have the potential to reduce habitat quality.

9.2.2 Recommendations and Research Needs

(1) Any habitat inventory should take into account small temporary and semi-permanent sloughs which are of greatest importance to nesting waterfowl. These inventories should identify surrounding land use in detail (ie. major crop type) as this may have a bearing on the pressures exerted on the wetland. Also, these inventories should be repeated at regular intervals.

(2) The CLI inventory was found to be useful for predicting waterfowl breeding densities and should be repeated and compared to the earlier one, especially the "actual" land use subset which dates back to 1967. Since the presence of permanent or semi-permanent wetlands was a key factor in establishing waterfowl land classes, and since these may not be the most attractive to nesting ducks, more validation of the CLI waterfowl classification should be undertaken before repeating the inventory. Priority should be given to surveys in Saskatchewan where there is the greatest potential for conflict between agricultural and waterfowl resource interests.

(3) In order to improve estimates of current loss of waterfowl to agriculture and the role of pesticides in contributing to these losses, the annual waterfowl transect data should be analyzed in light of the agricultural use of surrounding lands. Factors such as crop type, cultivation practices, the extent of slough tillage and clearing as well as pesticide use should be incorporated. This will require that farmers whose lands fall on the established transects be contacted directly for this information. As there is evidence that air ground transects, on account of their placement near roads, are atypical of general prairie conditions, transects other than these should be established for monitoring the loss of wetlands.

(4) Research is needed to confirm the relative importance for duck nesting of forage and other crops and to advise farmers on planting and cropping practices which optimise cover crop use by nesting waterfowl.

(5) A program of incentives and/or education should be developed to protect sloughs and slough borders from the impacts of pesticides and other agricultural disturbances.

9.3 PESTICIDE USE IN THE PRAIRIES (CHAPTER 4)

9.3.1 Summary

There are several sources of information on pesticide use but, unfortunately, none are sufficiently detailed for use in environmental planning or impact assessment. The best data are either deemed to be confidential business information (e.g. registrant surveys), pooled by several actives (e.g. Daciw, annual), geographically restricted (e.g. Manitoba crop insurance surveys) or incompletely reported (e.g. Canadian Agricultural Insect Pest Reviews). The reporting is generally poorer for insecticides than for herbicides.

The prevalent insecticide in the Canadian prairies is carbofuran (Furadan®). For grasshopper control, it currently ranks number one, followed by deltamethrin (Decis®) and carbaryl (Sevin®). Dimethoate (Cygon®), which has been the main replacement for dieldrin since the late 1960's, has been phased out except for a registered bran-bait formulation for control of grasshoppers. However, this active may be revived if new odourless formulations are introduced. Chlorpyrifos (Lorsban®) is expected to start competing successfully for use in grasshopper control because of new, lowered application rates. Malathion is heavily used but it is difficult to separate its use in field crops from its use against woodland pests (tent caterpillars), biting flies (equine encephalitis programs) and its use on stored grain. Two immediate regulatory problems appear to be the illegal use of bran baits and the mixing of insecticide cocktails by farmers.

Grasshoppers are the most severe pests of prairie Canada, although their numbers fluctuate greatly from one year to the next. Based on data from the last decade, approximately two million hectares are sprayed for grasshoppers in a moderate-severe outbreak year. The extended area (including reapplications) of spray during the 1985 outbreak was estimated to have been between 3.7 and 4.7 million hectares. Spray areas could have been substantially higher had better growing conditions prevailed during that summer.

The area of spray cannot be predicted from infestation forecasts. A comparison of the two shows an uncertainty spanning three orders of magnitude. Although grasshoppers tend to be associated with dry conditions, the relationship between the presence of water and years of heavy spraying is not clear. During two recent heavy-spray years (1974 and 1982), the number of July ponds, as determined by the annual waterfowl surveys, were almost at a record high in the areas of southern Saskatchewan most affected by grasshoppers. Spraying may, in fact, be relatively heavy in wet years since farmers then have a greater expectation of return on their crop.

It is very difficult to estimate the extent of aerial spraying for grasshoppers. "Educated guesses" by provincial agriculture officials vary from a low of 10% of the total area sprayed in Saskatchewan to 25%, in Manitoba. A weighted prairie average would place the proportion of aerial grasshopper spraying at 13% or less. In contrast to this low proportion of aerial spraying, pesticides were applied entirely by air during the largest recorded outbreaks for pests of canola (400,000 ha for the 1972 bertha armyworm infestation and 480,000 ha for the 1985 diamond-back moth infestation).

Based on recent data from Statistics Canada and Environment Canada, the "average" insecticide spray areas in years without major pest outbreaks were estimated to be in the range of 1.0 million ha sprayed at least once and possibly an extended area of up to 1.5 million ha if repeat spraying is included.

"Baseline" insect control programs (excluding seed treatments) were estimated to take place on 3-4% of improved agricultural land. A serious grasshopper outbreak would increase this proportion to 8-9%. A combination of a grasshopper outbreak with an outbreak of a major pest of canola would increase the spray area to 10-11% of improved lands. In 1985, a remarkable year, 6.5-7 million hectares were sprayed with insecticides, an area which corresponds to 17-19% of all agricultural "improved" lands or 21-23% of the best cropland (land classified as Agriculture Classes 1-3). Temporally, all of these spray programs overlap to some extent with the duck breeding season. Sprays of canola pests tend to be later in the season and may overlap to a larger extent with nesting by diving ducks.

As stated earlier, the reporting of herbicide use is easier to quantify since it tends to be more static than insecticide use both in time and space. The most important recent change in herbicide use has been the increased control of wild oats, especially in the parklands. It is estimated, by a variety of methods, that approximately 13-15 million hectares are sprayed yearly with grassy or broadleaf herbicides. Some crops are sprayed extensively. For example, an estimated 80% of the wheat crop is sprayed with broadleaf herbicides and 91.5% of the canola crop is sprayed with grass herbicides. For broadleaf control, 2,4-D in amine or low-volatile ester form is still the most widely used herbicide. However, there has been a steady increase in the use of MCPA since mid-1960's and MCPA use is now 2/3 of 2,4-D use. Bromoxynil (Pardner®) is increasingly used against 2,4-D- and MCPA-tolerant weeds. Triallate (Avadex®) is currently the herbicide of choice for wild oat control, followed closely by diclofop methyl (Hoe Grass®). Trifluralin (Treflan®) is heavily used in canola to control both wild oats and broadleafs in that crop.

Spring-applied sprays still represent the bulk of the herbicide market although fall-applied granulars are gaining in popularity. It is generally recognized that the gradual (but currently very slow) shift to reduced-till crop practices will result in both a quantitative and qualitative shift in herbicide use. Since reduced-tillage favours perennial weeds, new broad-scale, fast-acting but low-residual herbicides (such as paraquat (Sweep®) and glyphosate (Roundup®) are expected to gain in popularity. New post-emergence products such as fluazifop-butyl (Fusilade®), sethoxydim (Poast®) and clopyralid (Lontrel®) are seen as breakthroughs for reduced tillage strategies.

9.3.2 Recommendations and Research Needs

(1) To make pesticide use data more generally available for analysis, we recommend the following measures:

a) Removing the confidentiality status of registrant data on pesticide sales.

b) Extending and publishing Alberta figures on the use of pesticides by licensed applicators; separating aerial use from ground use.

c) Establishing some sort of pesticide use survey in Saskatchewan. This province has the highest use of pesticides and yet has the least information on use.

d) Standardizing the reporting of pest infestations in the "Canadian Agricultural Insect Pest Review" to provide information on estimated size and location of outbreaks, size of spray area, frequency and method of treatment as well as insecticides of choice. A fixed tabular format would be the most suitable for this purpose.

e) Modifying the Statistics Canada agricultural surveys to clearly separate insecticide spray from seed treatment and also insecticide use from fungicide use. It should also be made clearer whether numbers refer to one-time spraying or repeat spraying. Finally, farmers should be queried on pesticide use during every survey (i.e. every five years) rather than every second survey.

(2) To stem current abuses in pesticide use, we recommend that regulatory authorities:

a) Crack down on open endorsements by the agricultural research community of unregistered pesticides or formulations.

b) Crack down on the use of insecticide cocktails until research can demonstrate their safety.

(3) In order to better define the impact of grasshopper spraying on the waterfowl resource, the relationship between pond number, moisture conditions and grasshopper spraying should be investigatged.

(4) From an agronomic research point of view:

a) There should be serious consideration given to the wisdom of growing cereal crops in areas with a limited expectation of return on account of climate and grasshoppers.

b) Spray programs in these areas should be critically assessed for their cost-effectiveness and an attempt should be made to calculate the true social and environmental costs of these programs.

9.4 LIKELIHOOD OF PESTICIDE CONTAMINATION OF WETLANDS (CHAPTER 5)

9.4.1 Summary

In Chapter 5 we examined the relative importance of the two major pesticide application methods (aerial and groundrig), the factors affecting off-target contamination and pesticide mobility, and the major routes of contamination of waterfowl breeding habitat in the Canadian prairie pothole region. Although the estimates of the amounts of pesticides sprayed aerially are subject to many uncertainties, the data available suggest that in 1980, 9% of the 12.6 million hectares sprayed with herbicides, and in 1985 over 13% of the 7 million hectares sprayed with insecticides, were treated by air.

The greatest potential source of contamination immediately following spraying is drift, either as droplets or vapour. Although highly variable, off-target drift of more than 50% of amounts applied is not uncommon for both aerial and groundrig application especially, in the latter case, when mist blowers are used. The major factors affecting drift are droplet size, wind speed and atmospheric stability. The smaller the droplet size the longer it will drift; droplet size is dictated by a compromise between the need for better pest control (small droplets) and the need to maximize on-target deposition (larger droplets). Winds with speeds high enough to produce significant drift are common in the prairies. Atmospheric stability plays an important role in the prairies as inversions are a common diurnal feature and act to maintain a shallow concentrated layer of spray and vapour drift near the ground which may travel horizontally for long distances. Other factors which increase drift are high temperatures and low relative humidity; properties of the final tank mixture such as low surface tension, low viscosity and volatile solvent fraction, decrease droplet size thereby increasing drift. Finally, rolling and rough terrain, typical around sloughs, act to slow groundrigs and increase wheel bounce which further increases the amount of off-target drift.

Direct routes of contamination such as overspray of ponds and off-target drift are the major pathways of pesticide entry into waterfowl breeding habitat (i.e. potholes and nesting habitat). Aerial overspray of ponds is almost impossible to avoid when the potholes are small or found in the high densities characteristic of many areas of the prairies. Similarly, when groundrigs spray the margins of sloughs it is likely that the boom will swing over at least part of the pond.

Drift droplet deposition away from the target area following either aerial or groundrig application can be substantial. Deposition of 20 to 40% of amounts applied within 100 m downwind of the target is a reasonable average. A large number of upwind swath applications will lead to greater deposition.

Vapour drift of material on the ground which evaporates hours following spraying can be substantial. Volatilization losses as high as 40% have been reported for certain forms of 2,4-D.

While surface runoff of pesticides can be high following heavy rains, these events are rare and very localized during the summer in the prairies. Eighty-five per cent of the runoff occurs during the spring snow-melt before pesticides are applied.
The potential for leaching is greatest for carbofuran, carbaryl, 2,4-D amine, dicamba, amitrole, dalapon, triazines, paraquat and picloram. Of potentially greater concern is the leaching of pesticides into groundwater followed by discharge into sloughs. However, this route of contamination still needs to be investigated.

We concluded that the possibility of contamination of non- target areas following the application of pesticides is very high regardless of the application method. This is especially true of habitats contiguous with the sprayed fields. The amounts of pesticides contaminating such habitats will be determined in large part by local meteorological and topographical conditions; under certain conditions the amounts reaching off-target sites may equal or even exceed those reaching the target.

9.4.2 Recommendations and Research Needs

(1) Accurate annual information on the amount of pesticides applied aerially or by groundrig is lacking; annual surveys of all aerial applications should be done in conjunction with surveys of all pesticide use in the prairies (see Section 9.3.2). Also, the factors which determine the quantities of pesticides sprayed aerially should be clearly identified and their effects quantified (e.g. soil moisture in spring, application costs, levels of infestation).

(2) Although much is known about the behaviour of drift and the factors affecting it, information is lacking on the role that could be played by buffer zones in reducing off-target contamination. The size of such zones and their compostion and effect on drift should be investigated in the context of the agricultural landscape of the prairie pothole region.

(3) Water monitoring agencies such as Inland Waters Directorate should be encouraged to monitor pesticide residues in slough waters.

9.5 TOXICITY OF INSECTICIDES IN SLOUGH ECOSYSTEMS: INDIRECT THREAT TO WATERFOWL (CHAPTER 6)

9.5.1 Summary

Chapter 6 began with a review of the relative toxicity of major use insecticides to aquatic invertebrates and of model ecosystem and field studies of insecticide impacts on aquatic ecosystems. Inferences were then made about the hazard posed by insecticide contamination of sloughs to breeding waterfowl.

There is much variability in the response of aquatic invertebrates, even among those belonging to the same taxonomic group, to acute exposure to insecticides. While the toxicity data are variable and not always comparable, insecticides can be pooled into groups of similar toxicity. An overall review of the available data showed that deltamethrin and cypermethrin are the most acutely toxic to invertebrates (24h LC_{50} = 0.02 to 0.9 μg L^{-1}). The next most toxic group consists of permethrin, fenvalerate and chlorpyrifos (24h LC_{50} = 0.5 to80 μg L^{-1}). These are followed by malathion, diazinon and methoxychlor (24h LC_{50} = 15.0 to 100.0 μg L^{-1}). Finally, carbaryl is the least toxic of those compounds for which there was enough comparable data (24h LC_{50} = 75.0 to 650 μg L^{-1}).

The pyrethroids are toxic to most arthropods except for molluscs. Dipterans and ephemeropterans appear to be the most sensitive taxa.

Little work has been done on chronic toxicity of insecticides to aquatic invertebrates, although there is evidence that the more hydrophobic compounds accumulate more readily in organisms, possibly leading to long-term chronic toxicity.

Field studies of insecticide impact on aquatic ecosystems consist of microcosm and enclosure studies, controlled toxicity tests in ponds, sloughs and lakes, and monitoring studies of aquatic systems following insecticide spraying for pest or vector control. The large differences in test conditions, test ecosystems and sampling techniques make detailed comparisons of the severity and duration of impact difficult. The available data suggests that synthetic pyrethroids, azinphos methyl, chlorpyrifos, malathion and, to a lesser extent, carbaryl and methoxychlor have substantial impacts on aquatic ecosystems at rates recommended for control of cereal crop pests. The pyrethroids decrease invertebrate species diversity and abundance substantially. There is little recovery of invertebrates for up to four months and, in some cases, for up to one year. Most taxa of invertebrates are affected except for oligochaetes and molluscs. For the other major-use insecticides, the extent of the impact varies substantially. The responses of the aquatic communities range from a simple change in the dominant invertebrate species to the indiscriminate die-off of most invertebrate life along with long recovery times. Multiple applications, closely spaced in time, can adversely affect even tolerant populations and dampen the resilience of all exposed populations. Recovery is influenced by the availability of colonizers as well as the life history of the species affected; those invertebrates with short generation times will bounce back more quickly (e.g. zooplankton).

An ecological hazard evaluation of the major insecticides was conducted. Information on the environmental behaviour, accumulation patterns and toxicity of the insecticides were used to predict where a compound would likely be and for how long (exposure). The evaluation also suggested which taxa, communities and ecological processes would most likely be affected. Using computer models of the ecosystem, where the major physical and chemical components, the biota, and the ecological processes of the system are represented, the magnitude and persistence of expected pesticide concentrations were determined and compared to concentrations toxic to invertebrates. This exercise was carried out using three models. Firstly, a four-compartment (water, sediment, biota and a catchall [i.e. algae and suspended solids]) persistence model was used to predict the compartmentalization over time of insecticides sprayed directly over a pond under two pH conditions. Secondly, an acute exposure model was used to derive a relative hazard index which assesses the impact of insecticides to the aquatic fauna during the initial phase after spraying when concentrations are high but usually decreasing rapidly. Thirdly, a chronic exposure model was used to determine the same relative hazard during the second phase when concentrations are low and loss rates slow.

The persistence model predicted that pyrethroids are mainly sequestered in sediments (95%) after equilibrium has been established. Carbaryl, carbofuran, dimethoate, malathion and phosmet remain primarily in water while azinphos methyl, chlorpyrifos and methoxychlor are found mainly in sediments. Diazinon partitions nearly equally between the water and sediments. Except for fenvalerate, the pyrethroids have the longest system half-lives, mostly exceeding 1,000 days. At pH 5, the half-life of carbofuran exceeds 1500 days. The other compounds have half-lives which are usually less than 500 days. Carbaryl, diazinon and phosmet have half-lives of less than ten days. Except for methoxychlor, most compounds were less persistent at a higher pH.

The hazard to aquatic invertebrates predicted by using the exposure models is a function of the toxicity and the availability of the insecticides in the aqueous phase. The synthetic pyrethroids along with azinphos methyl, chlorpyrifos, malathion and methoxychlor fall into the high hazard category based on the acute exposure model. There are, however, some notable taxonomic differences in sensitivity. The ranking of certain compounds, such as malathion and diazinon, depends on pH.

The relative hazard ranking obtained under the chronic exposure model is substantially different from that obtained under the acute exposure model, and is much more sensitive to pH. Under this model, carbofuran and malathion join the pyrethroids, azinphos methyl, chlorpyrifos and methoxychlor in the high risk group. The long-term hazard posed by carbaryl, diazinon and dimethoate is relatively low.

From the results of the models and field studies, we predict that cypermethrin, deltamethrin, permethrin, azinphos methyl and chloryrifos will destroy a large proportion of the invertebrate populations in ponds and shallow lakes, regardless of pH, at recommended application rates. Population recovery will be slow for most taxa, particularly for those that do not have sources of mobile colonizers. Malathion is predicted to induce effects similar to permethrin under acidic conditions. Diazinon and dimethoate present little or no risk, with effects being expressed primarily through changes in community structure. While carbofuran may pose little risk under an acute exposure scenario, the lack of toxicity data for this compound limits the confidence in the ranking based on the model. Compounds like methoxychlor, with relative hazard values equivalent to carbaryl or slightly greater, are expected to produce a substantial and rapid die-off of sensitive invertebrates when shallow slough ecosystems are exposed at or near recommended application rates for crop protection. However, the duration of impact will be moderate (3-6 months) and some species important to duckling feeding (e.g. chironomids) may not be affected.

A duckling growth model was developed in order to predict the effect of insecticide depletion of slough macroinvertebrates on ducklings. Duckling dependency on animal protein, which is critical for rapid growth and therefore survival in the initial two to four weeks after hatching, was discussed in Chapter 2. There is evidence that the rate of growth of ducklings is a function of macroinvertebrate abundance in ponds: ducklings feeding on insecticide-treated and acid-stressed ponds had slower rates of growth than those on control ponds. The duckling growth model predicts the macroinvertebrate standing crop required to support various levels of growth (optimal, marginal [25% of optimal] and zero growth) for Mallard and Lesser Scaup ducklings. The model is based on two components: first, the relationship between age, live weight, dry matter intake and optimal growth of ducklings (after making assumptions about the efficiency of converting dry weight of food into live weight of duckling); and second, the dietary intake required to meet specified growth requirements as related to the invertebrate biomass available. This second component requires information about time spent feeding, feeding success per unit time, consumption rate per prey density and energy-biomass conversions for macroinvertebrates.

Relating predictions of necessary macroinvertebrate biomass from the duckling growth model to insecticide stress may be crudely done with the benchmark compounds carbaryl and permethrin for which there are a relatively large number of field studies. Assuming an average macroinvertebrate standing biomass of approximately 4.0 g dry weight m^{-3} in prairie potholes, insecticides of moderate hazard (equivalent to that of carbaryl) are expected to reduce standing crop levels of macroinvertebrates to less than 1.0 g dry weight m^{-3}. This prey resource level is predicted to be insufficient to support marginal growth in an average brood of six Mallard or Lesser Scaup ducklings. Compounds calculated to have a high relative hazard (equivalent to that of permethrin) are expected to deplete macroinvertebrate standing stocks by >90%, reducing expected biomass to <0.4 g dry weight m^{-3}. This biomass level is predicted to be just sufficient to support marginal growth in a single Lesser Scaup or Mallard duckling. It is totally inadequate for a normal size brood of one week of age.

9.5.2 Recommendations and Research Needs

(1) Laboratory toxicity testing:

a) more standardized toxicity data are necessary to enable a comparison of insecticides;

b) test species should be consistent and representative of those which are important to waterfowl and fish,

c) more studies of delayed mortality, speed and reversibility of toxic response, and the effect of short exposure periods are needed.

(2) Field studies:

When compared to laboratory studies, there are even greater differences in test conditions, test ecosystems and sampling techniques in field studies. This makes detailed comparisons of the severity and duration of the impact of insecticides on aquatic systems very difficult. For the most part, the field studies available do not properly address sampling and statistical criteria essential to the quantification of community change. We recommend therefore:

a) the establishment of a standard pond field study protocol which would allow comparisons among insecticides;

b) that the experimental design also addresses long-term effects;

c) that some work be carried out on ecosystems which are most likely to be contaminated under real use conditions;

d) that the sampling design take into account the large variability among pond systems;

e) that the studies also focus on the indirect impact on resources (e.g. food for vertebrates, habitat).

(3) Benchmarks and models:

a) there is a need to develop better data sets on field studies of benchmark compounds such as carbaryl and permethrin and to establish more benchmarks against which other pesticides, especially new ones, can be quickly evaluated;

b) there is a need to develop better models of pesticide fate and effects, especially for benchmark compounds; this would enhance the predictive power of models and make them more useful for pesticide evaluation.

9.6 TOXICITY OF HERBICIDES IN SLOUGH ECOSYSTEMS AND UPLAND HABITAT: INDIRECT THREAT TO WATERFOWL (CHAPTER 7)

9.6.1 Summary

In this chapter we first examined the factors which control the distribution and persistence of herbicides in sloughs and their margins. This is followed by a review and analysis of the impact of herbicides on the foods and vegetative habitat of prairie waterfowl.

As prairie sloughs generally have sediments rich in organic matter and relatively dense macrophyte vegetation, adsorbent surfaces may be more available than in agricultural soils. The expected equilibrated distribution of selected herbicides in contaminated sloughs is summarized based on soil and aquatic partitioning studies and inferences from structurally similar compounds. The strong and weak acids (2,4-D acid, dalapon, dicamba, dichlorprop, MCPA, mecoprop, TCA) are expected to remain primarily in the water, with strong adsorption of weak acids occurring only below pH values equal to pKa. The triazine compounds (atrazine, cyanazine, metribuzin, simazine) are expected to be divided between water, sediments and macrophytes. The cationic bipyridilium compounds (diquat, paraquat), nonionizable hydrophobic compounds (linuron and trifluralin) and the less soluble carbamates (triallate) are expected to partition almost exclusively to the sediments and macrophytes.

The persistence of herbicides in slough ecosystems is determined primarily by their rate of chemical, photochemical and biological degradation. These rates are influenced by soil type, soil-water content, pH, temperature, clay and organic matter content of the soil. The most profound yet highly unpredictable rate-controlling factor is microbial activity. In flooded soils or deep slough sediments, herbicide degradation rates depend on the relative activity of aerobic versus anaerobic microbes.

Based on the very limited data available for aquatic persistence of the major-use herbicides, the following classification is suggested as a basis for chronic exposure assessment. In the persistent category ($t\frac{1}{2} > 100$ days), diquat, paraquat and linuron should be assessed for their potential chronic effects. Under aerobic conditions, trifluralin is also highly persistent. Trifluralin, triallate and picloram are included within this persistent group because of documented residue carry-over in agricultural fields. The triazines are moderately persistent ($t\frac{1}{2} < 100$ days) but should be considered a potential chronic threat to slough biota because of their high degree of availability in water and their relatively high toxicity to plants. Also included in the moderately persistent group are glyphosate and compounds for which volatilization losses may decrease persistence: EPTC, barban, asulam, propanil, trifluralin (anaerobic conditions). Finally, the least persistent compounds ($t\frac{1}{2} < 20$ days) consist of the acids 2,4-D, MCPA, dichlorprop, mecoprop, dicamba, dalapon and TCA.

Herbicide effects on slough and upland vegetation can have a negative impact on waterfowl through reductions in plant foods and nesting cover. The response of algae to herbicides is highly variable between compounds for any given species of alga and between algal species for any given compound. Also, certain herbicides will stimulate algal production at very low concentrations while inhibiting it at high concentrations. Field studies indicate that the triazine compounds are the most potent inhibitors of phytoplankton growth at the acute levels of exposure expected after direct overspraying of ponds. Recovery times, however, are rapid (one to three weeks).

Little is known about the effect of herbicides on macrophytes other than from the monitoring of the impact of chemical treatment of water bodies for aquatic weed control. A review of the literature indicates that 2,4-D amine, atrazine and metribuzine, sprayed on ponds at rates recommended for crop weed control, may result in the die-off of macrophytes. Chronic exposure of submergents to low levels of the butoxyethanol ester (BEE) of 2,4-D and diquat led to 90% mortality. Therefore, compounds which combine high phytotoxicity and high persistence such as diquat, paraquat and the triazine herbicides may have a significant impact on wetland vegetation when used on a yearly basis.

The die-off of macrophytes following treatment with herbicides generally leads to decreases in dissolved oxygen, pH and concentrations of carbonate ion concentrations and increases in free CO_2, bicarbonate and alkalinity. Substantial inputs of nutrients (N and P) often follow such events and can lead to phytoplankton blooms. Most macrophytes recover within one year except for some emergents (*Typha*, *Sparganium*, *Phragmites*) which may take up to three years.

In the prairie agricultural landscape, the habitats which still provide good nesting cover consist mostly of roadsides, slough margins, rights-of-way, hay fields and idle land. Because of their small size and isolation, maintenance of abundant plant cover within these habitats is of great importance to duck production. While herbicide effects on weed populations in cultivated fields are relatively well researched, their impact on natural plant communities in the prairies has not been investigated. However, there have been a number of studies of natural plant assemblages (i.e. non-crop) sprayed with herbicides for management purposes (e.g. forestry, roadside maintenance, utility right-of-way).

While the initial effect of herbicide application is a reduction in abundance of undesirable plant species, the result is often variable and depends on the original species composition of the habitat, the type of plant targeted, the type of herbicide applied and the frequency and selectivity of its application. Herbicide use usually leads to a decrease in plant diversity and the establishment of resistant species. The use of selective herbicides causes interspecific shifts in the population. These changes in species composition are not always accompanied by changes in total plant biomass. Of greater relevance to breeding ducks are changes in the amount of cover provided by the vegetation. The use of phenoxy herbicides is known to reduce average vegetation height by as much as 50%. Also, broadleaf herbicides cause a shift in the forb:grass ratio in favour of grasses. These effects most likely reduce the amount of that cover which is preferred by breeding hens for nesting. While little or no work has been done on herbicide effects on the upland habitat which provides good nesting cover, the fact that these habitats are close or adjacent to fields under cultivation suggests that contamination by herbicide drift probably occurs and has some impact. The consequences to ducks are difficult to assess, although one study did report that 2,4-D applications to prime nesting habitat on an island in Alberta resulted in a 75% decline in nest densities.

Herbicides, in general, are not as toxic to aquatic invertebrates as are insecticides. The LC_{50} values for the most toxic herbicides (2,4-D esters, triallate, trifluralin, atrazine and simazine) are one to two orders of magnitude less toxic than are the more toxic insecticides to the same species. Only 2,4-D BEE has an LC_{50} to Daphnia lower than concentrations expected in sloughs after direct overspray, although the ester hydrolyses rapidly. As a group, zooplankton appear most susceptible to herbicide toxicity. However, as a prey group, they are of minor importance in the diet of ducks, with the exception of Shovelers. Amphipods are at great risk from diquat in its non-adsorbed state. Because of their habits, amphipods may be further exposed to relatively high and stable diquat concentrations during feeding on contaminated sediments and aquatic plants. Insects, as a group, are most sensitive to the triazine herbicides, particularly atrazine, which has chronic effects on midges. The 2,4-D esters, atrazine, simazine and trifluralin, can pose a consistent and moderate hazard to the major invertebrate taxa which contribute substantially to duckling feeding. However, the severity of hazard they posed to the aquatic invertebrate community, based on the analysis of laboratory direct toxicity data, is relatively minor in comparison to that posed by the highly toxic insecticides.

There is little evidence from field studies that applications of herbicides to shallow lentic water bodies, at normal crop protection rates, have direct toxic effects on aquatic invertebrate populations. However, because of the close association between macrophytes and invertebrates, the primary impact of herbicides on the invertebrate fauna is related to the loss of macrophyte habitat. The elimination of aquatic plants most often results in severe decreases in gastropods and most orders of crustaceans and aquatic insects. This is not surprising in light of the fact that the average number of macroinvertebrates associated with vegetation in ponds has been found to be approximately ten times the abundance on the bottom or in the water column. The only taxa which may increase in numbers after the die-off of macrophytes are detrital feeders, such as oligochaetes and larvae of some species of midges which respond to the increase in dead plant material, and zooplankton which feed on phytoplankton "blooms" which follow such die-offs.

It is clear that the hazard posed to ducks by herbicide use in the prairies is related to habitat destruction through the elimination or modification of upland and slough vegetation.

9.6.2 Recommendations and Research Needs

(1) There are few data on the environmental behaviour of herbicides in aquatic ecosystems; most studies on fate and persistence in the environment were conducted in agricultural soils. There is a need for comparative work on the behaviour of herbicides in aquatic ecosystems; such information would facilitate the process of pesticide evaluation for those concerned with conservation and protection.

(2) There is an important need for work on the direct and long-term chronic toxicity of major-use herbicides on slough vegetation and nesting cover which forms a patchwork mosaic in the agricultural landscape of the prairies; this kind of work is critical in light of the abundant yearly use of herbicides in the prairies.

(3) While there is little indication of serious acute toxicity of herbicides to invertebrates, a need to examine chronic effects of the more toxic and pesistent herbicides exists.

9.7 HAZARD EVALUATION (CHAPTER 8)

9.7.1 Summary

Of the 16 million ducks (26% of the total North American duck population) arriving on the Canadian prairies in the spring to reproduce, slightly more than 10 million nest in intimate association with agricultural cropland, either in the crops proper or, more commonly, along field edges, slough borders, fencelines, etc. As a result of agricultural encroachment, suitable nesting and feeding habitat is disappearing at an alarming rate. In this context the newer more effective insecticides and herbicides, ubiquitous in today's agricultural landscape, pose an indirect threat (i.e. ecologically mediated) to waterfowl production. The purpose of Chapter 8 was to determine the extent to which these impacts are being realized today.

While not reviewed specifically in this report, the acute toxicity of certain insecticides to birds is of some concern. Of particular note is carbofuran, which is used heavily for grasshopper and flea-beetle control and is also registered for use against most other prairie insect pests.

Several insecticides have the capacity to devastate the invertebrate community of prairie sloughs to such a degree as to render the pond incapable of supporting duckling growth for that season (see Chapter 6). We attempted here to determine how many birds are likely to be affected on account of their association with agricultural land. Using various assumptions and data sets on population size, nest density and success by habitat type and crop distribution, we apportioned the duck population and nests by census district, crop and habitat type. Knowing which insecticide is used for each pest and the proportion of each crop type sprayed, we could therefore estimate the number of birds likely to be affected. This exercise was done for three scenarios: one for annual insecticide use in non-outbreak situations, one for years of major grasshopper outbreak and one for an outbreak of a major canola pest.

In the first case we estimate that 0.32 million dabblers (2.3% of Canadian prairie dabblers) nesting in crops will be sprayed directly, with another 0.13 million nesting in margins exposed to drift. Under our scenario conditions 50,000 dabbler broods (or 3% of dabbler production associated with cropland) would be exposed. Forty-five thousand adult divers (or 3.4% of the diver population) and their broods would be nesting within areas sprayed with insecticides. Direct impacts are especially likely where aerial application is involved, an estimated 13% of the birds would be exposed. This corresponds to about one-half of one percent of the dabbler and diver production associated with cropland.

In worst-case (1985) grasshopper outbreaks we estimate that 16%, 5.3% and 20% of the infested land area will be sprayed in Saskatchewan, Alberta and Manitoba, respectively (or 2.8, 0.53 and 0.30 million ha treated), 13% of which will be sprayed aerially. We therefore calculated that some 0.93 million dabblers and 0.13 million divers (6.9% and 6.2% of their respective Canadian prairie populations) would be within the spray areas. In 1985, half of the total spray was carbofuran which represents a direct acute hazard for 0.46 million nesting dabblers and their broods. Indirect impacts via reduction of the invertebrate food supply of ducklings were likely with deltamethrin which represented 22% of the total insecticides applied in 1985 and where aerial application was used. Therefore, the habitat of 27,000 dabblers and 3,700 divers might not have supported any growth of duck broods. In short, a major grasshopper spray program is likely to affect anywhere from 30,000 to 1.06 million breeding ducks either directly or indirectly.

Given a scenario of a late (end of July-August) spray program for bertha armyworm or diamond-back moth control, only the population of diving ducks nesting in association with wetlands in the canola crop is at risk. On this basis we estimate that 43,000 divers (or 2% of the Canadian prairie population) will be at risk from such a spray program. Direct overspray is almost assured given 100% aerial application. If direct impacts occur, these are likely to be found in those areas sprayed with methomyl which is the most acutely toxic of the insecticides used. Indirect impacts are most probable with deltamethrin and chlorpyrifos and are also likely with malathion and methomyl.

Finally, to sum up the evaluation of hazards posed by insecticides, grasshopper control programs have the potential for the greatest impact on waterfowl, in terms of magnitude. While the percentage of ducks (usually 2-7%) which may be at risk from insecticide use is small for any one scenario, the scenarios are not mutually exclusive but are additive. All three scenarios occurred in 1985. Furthermore, any impacts resulting from insecticide use are added to impacts stemming from other agricultural disturbances. As the availability of the nesting and feeding habitat continues to shrink, the quality of the remaining habitat will assume a greater importance. With respect to their potential for an indirect impact via their toxicity to aquatic invertebrates, the synthetic pyrethroids cypermethrin, deltamethrin and permethrin, along with azinphos-methyl and chlorpyrifos, pose the highest hazard to duck populations. The compounds carbaryl, methoxychlor, carbofuran and malathion can pose a moderate to high hazard depending on environmental conditions.

The number of ducks affected by insecticide use may be underestimated if (1) aerial application becomes more popular, (2) deltamethrin and other insecticides of high invertebrate toxicity become more commonly used, and (3) synthetic pyrethroids and other compounds of high invertebrate toxicity but low vertebrate toxicity become the material of choice for aerial delivery.

Of the 310,315 km^2 of land under crop or summer fallow in the grassland-parkland region of the Canadian prairies, approximately 42.4% and 33.2% of the area is treated annually with broadleaf and grass herbicides, respectively. A review of herbicide toxicity to aquatic invertebrates and slough and upland vegetation (Chapter 7) indicated that, in light of the diminishing amount of upland nesting habitat within the agricultural landscape and because the presence of adequate plant cover is crucial for successful nesting, hatching and brood rearing, the greatest risk posed to ducks by herbicides is the reduction in the quality and quantity of cover.

Many activities which take place in the field proper are destructive to duck nests and habitat; therefore, herbicide applications will probably add little to the disruption of the natural system already taking place. Thus, the hazard assessment of herbicides focused on the effects of the contamination of those habitats which still provide conditions suitable for successful reproduction of waterfowl such as slough margins, idle grassland, hay fields, rights-of-way, roadsides or headlands.

To assess the extent of the hazard posed by herbicides, we calculated the production of successful nests by prairie dabblers (upland nesters) under various scenarios of reductions in nest densities and success in the "margins" (roadsides, idle land, headlands, slough edges, etc.) which would be expected to occur if these habitats were treated with herbicides. These estimates were then compared to data from field studies. To carry this exercise through, we used the distribution of ducks by census district, crop type and habitat type previously determined for the evaluation of insecticides assuming the following: (1) that the sex ratio is 1:1, (2) that, in determining the percentage of each crop sprayed, the use of grass herbicides was independent of the use of broadleaf herbicides, (3) that "margins" which provide the best nesting habitat constitute 10% of the land area, and (4) that all hens will initiate only one nest during the season (hen success = nest success).

Calculation of the baseline production of successful nests (i.e. no reductions in nest success or densities) shows that the margins contribute 60.4% of all successful nests in spite of making up only 10% of the total surface area. This large contribution of margins is easily accounted for by nest densities and success rates, three and four times greater, respectively, than in the crop itself. Crop production of nests is as high as 39.6% because of the large contribution of tame hay where nest densities and success rates are equal to those of the margins; if tame hay production is ignored, total crops contribute just under 10% of all successful nests.

Under this scenario, reducing nest densities in the margins forced part of the hen population to nest elsewhere, while reducing nest success within the margins simply decreased the number of successful nests produced for the hen population nesting in that habitat. Reducing only nest success we find that there is a linear decrease in the number of successful nests produced up to the point where nest success equals zero and where successful nest production within sprayed margins is down to zero irrespective of the number of birds nesting within. The impact of reducing nest success is more severe at higher values of nest density; the lower the nest density is, the shallower the slope of the line relating nest success and successful nest production will be.

The maximum potential loss in production was estimated to be 30% when nest densities are left unchanged and nest success is reduced to zero in margins. It is, however, not likely that nest success will fall below the level recorded in field crops or that nest densities will stay unchanged after the vegetation is modified by herbicides. In the only field study of this problem, duck nest densities were reduced by 75% when broadleaf plants were removed from prime nesting habitat by herbicides. If we use this value to extrapolate to our calculations, we find that the decline in successful nest production will range from 7.3 to 17.6%.

In spite of the crudeness of the scenarios developed for this evaluation, the assessment leads us to believe that herbicide use has the potential to reduce successful nest production in agricultural areas of the Canadian prairie pothole region. In addition, this analysis was not expanded to include brood success as the fate of ducklings between hatching and fledging has been poorly studied. There are grounds to believe that reductions in vegetative cover would lead to decreased survival of ducklings during this period due to increased predation losses. Furthermore, because of a lack of information, this evaluation did not take into account the effect of long-term chronic exposure of slough invertebrates and plants to persistent herbicides.

9.8 CONCLUDING REMARKS

This report points to legitimate concerns about both the indirect and, to a lesser extent, direct impact of pesticides on waterfowl in the Canadian prairie pothole region. The indirect impact results from the toxicity of certain pesticides to aquatic invertebrates and vegetation in sloughs and uplands which play a dominant role in the successful recruitment of prairie ducks. We have clearly identified the pathways through which the pesticides have an impact and have estimated the number of ducks likely to be affected. While the numbers are only approximations, they indicate that the potential impact is greater than generally believed. Any impact only adds to other sources of duck mortality and habitat degradation. Greater pesticide use in the future, increased use of aerial application and the development of pesticides with greater toxicity to invertebrates or plants will only increase the potential for a negative impact on waterfowl.

Of great concern is the paucity of information available to carry out a more accurate evaluation of the hazard posed by pesticide use to waterfowl. The data gaps and research needs are identified in some detail in the preceding sections for each chapter of this report. These range from questions on the variability of primary and secondary production of prairie sloughs and how this is related to waterfowl use of these ponds and the concept of carrying capacity, to questions on pesticide use patterns and quantities applied, waterfowl censuses, pesticide drift and finally the environmental behaviour, especially on a long-term basis, of these compounds in aquatic systems.

While filling in the data gaps may give us more precise numbers, this may not be enough to solve some of the assessment problems encountered here. Part of the problem lies in the fact that there are two broad categories of data sets which are collected at very different scales: firstly, data on the toxicity and environmental behaviour of pesticides collected in field and laboratory studies and secondly, waterfowl census and pesticide use data for the prairies. In carrying out this assessment, it was very difficult to connect these two levels of information collection because of wide differences in spatial and temporal scales and in variability (e.g. controlled experiments vs. survey data). This gap in the gathering of information points to the need for more realism in the collection of data useful for hazard evaluation. For example, while we know how much pesticides are sold in Canada, we are not sure how and when they are used. In addition, there is little information on the distribution of duck habitat within farmland, on actual farming practices or on the attitude of farmers towards off-target pesticide contamination and wildlife. In short, the issue of pesticide impact on waterfowl has to be assessed within the larger context of waterfowl and habitat management, pest management, soil and water conservation and farm economics.

One particular study of considerable relevance to this problem is the Cereals and Game Birds Research Project, underway since 1983 in northeast Hampshire, England, which was "set up to quantify the effects of pesticides on farmland wildlife, including game birds, and to seek practical methods of conserving wildlife that are compatible with modern farming ..." (Rands & Sotherton, New Scientist, July 4, 1985, p. 32). This project, entirely funded by farmers, arose from a concern about the decline of the Grey Partridge and other wildlife dependent on farmland. The biologists and agronomists involved found that "a small reduction in pesticide use (less than 2% of the total field area) can go some way towards assisting the conservation of wildlife ..." without any noticeable changes in cereal yield.

Such a medium-scale study (50 to 100 km^2 in area) conducted on Canadian prairie farmland could go a long way in answering some of the questions raised in this report. The co-operation of ecologists, hydrologists, pedologists, toxicologists, chemists, agronomists, economists and farmers on such a study would allow us to tackle, with greater realism, the pressing problems of soil, water and habitat conservation and chemical contamination.

BIBLIOGRAPHY

A.A. Aquatic Research Ltd.. 1985. Implications of reduced and non-tillage systems on chemical use in the prairie provinces. Report prepared for Environment Canada. Environmental Protection Service, Western and Northern Region.

Abram, F.S.H., C. Eving and J.A. Hobson. 1980. Permethrin for the control of animals in water mains. Water Research Centre Technical Report, England.

Adams, R.T. and F.M. Furisu. 1976. Simulation of pesticide movement on small agricultural watersheds. EPA-600/3-76-066. U.S. Environmental Protection Agency. Athens, Georgia.

Agriculture Canada. 1984. Compendium of pest control products registered in Canada. Seed Dressings. Agriculture Canada Publication 1654 SD/81 (updated to 1984).

Ahrens, J.F. and P.M. Block. 1972. Phytotoxicity and persistence of simazine in a shallow lake. Proc. Northeast. Weed Sci. Soc. 26:166.

Akesson, N.B. and W.E. Yates. 1964. Problems relating to application of agricultural chemicals and resulting drift residues. Ann. Rev. Entomol. 9:285-318.

Akesson, N.B. and W.E. Yates. 1984. Physical parameters affecting aircraft spray application. In Chemical and Biological Controls in Forestry. ASC Symposium Series No. 238. pp. 95-115.

Alberta Agriculture. 1972. 1972 crop distribution in Alberta, Saskatchewan, Manitoba. Agdex 110-01. 46 p.

Alberta Agriculture. 1982. Guide to weed control in Alberta. Part 1. Chemical. Edmonton. 170 p.

Alberta Environment Conservation Authority. 1974a. Biocide use in Alberta. Part 4: The use of insecticides in Alberta. Edmonton, Alberta. 43 p.

Alberta Environment Conservation Authority. 1974b. Biocide use in Alberta. Part 3: The use of herbicides in Alberta. Edmonton, Alberta. 83 p.

Alberta Environment Conservation Authority. 1976. The use of pesticides and herbicides in Alberta. Report and recommendations. Edmonton, Alberta. 208 p.

Ali, A. and M.S. Mulla. 1978a. Declining field efficacy of chlorpyrifos against chironomid midges and laboratory evaluation of substitute larvicides. J. Econ. Entomol. 71:778-782.

Ali, A. and M.S. Mulla. 1978b. Impact of the insect growth regulator diflubenzuron on invertebrates in a residential-recreational lake. Arch. Environm. Contam. Toxicol. 7:483-491.

Ali, A. and M.S. Mulla. 1980. Activity of organophosphate and synthetic pyrethroid insecticides against pestiferous midges in some southern California flood control channels. Mosqu. News 40(4):593-597.

Ali, A., M.S. Mulla, A.R. Pfuntner and L.L. Luna. 1978. Pestiferous midges and their control in a shallow residential-recreational lake in southern California. Mosqu. News 38(4):528-535.

Allen, H.E. and J.R. Kramer. 1972. Nutrients in natural waters. Wiley, New York. 457 p.

Allen, H.L.. 1971. Primary productivity, chemo-organotrophy, and nutritional interactions of epiphytic algae and bacteria on macrophytes in the littoral of a lake. Ecol. Monogr. 41(2):97-127.

Allison, D.T.. 1977. Use of exposure units for estimating aquatic toxicity of organophosphate pesticides. U.S. Environmental Protection Agency. EPA600/3-77/077.

Aly. O.M. and M.A. El Dib. 1971. Studies on the persistence of some carbamate insecticides in the aquatic environment. I. Hydrolysis of sevin, baygon, pyrolar and dimetilon in water. Water Res. 5:1191-1205.

Aly, O.M. and S.O. Faust. 1964. Studies on the fate of 2,4-D and ester derivatives in natural surface waters. J. Agric. Food. Chem. 12:541-546.

Ambrose R.B. Jr., S.I. Hill and L.A. Mulkey. 1983. User manual for the chemical transport and fate model (TOXIWASP), version 1.. U.S. Environmental Protection Agency. Athens, Georgia. EPA600/3-83-005

Anderson, M.G.. 1978. Distribution and production of sago pondweed Potamogeton pectinatus on a northern prairie marsh. Ecology 59(1):154-160.

Anderson, R.L.. 1980. Chironomidae toxicity tests, biological background and procedures. In Aquatic Invertebrate Bioassays, A.L. Buikema Jr. and J. Cairns Jr. (Eds.). American Society for Testing and Materials. pp. 70-80.

Anderson, R.L.. 1982. Toxicity of fenvalerate and permethrin to several non-target aquatic invertebrates. Environ. Entomol. 11:1251-1257.

Anonymous. 1984. Dusters of a new era. International Flying Farmer June 1984:8-9.

Anonymous. 1985. North American waterfowl management plan. Draft, November 1985. Canadian Wildlife Service-U.S. Fish and Wildlife Service. 37 p.

Apperson, C.S., R. Elston and W. Castle. 1976. Biological effects and persistence of methyl parathion in Clear Lake, California. Environmental Entomology 5(6):1116-1120.

Argauer, R.J., H.C. Mason, C. Corley, A.H. Higgins, J.N. Sauls and L.A. Liljedahl. 1968. Drift of water-diluted and undiluted formulations of malathion and azinphosmethyl applied by airplane. J. Agric. Entomol. 61(4):1015-1020.

Armstrong, D.P. and T.D. Nudds. 1985. Factors influencing invertebrate size distributions in prairie potholes and implications for coexisting duck species. Unpublished manuscript.

Arsenovic, M., A. Stankovic, M. Dimitrijevic and B. Konstantinovic. 1982. Investigations into the control of submerged vegetation in an irrigation system by specific herbicides. In 6th International Symposium on Aquatic Weeds. EWRS, Novi Sad. pp. 177-184.

Arthur, J.W.. 1980. Review of freshwater bioassay procedures for selected amphipods. In Aquatic Invertebrate Bioassays, A.L. Buikema Jr. and J. Cairns Jr. (Eds.). American Society for Testing and Materials. pp. 98-108.

Arthur, J.W., J.A. Zischke, K.N. Allen and R.O. Hermanutz. 1983. Effects of diazinon on macroinvertebrates and insect emergence in outdoor experimental channels. Aquat. Toxicol. 4:283-301.

Arvik, J.H., D.L. Hyzak and R.L. Zimdahl. 1973. Effect of metribuzin and two analogs on five species of algae. Weed Sci. 21(3):173-175.

Asher, S.C., K.M. Lloyd, D. Mackay, S. Paterson and J.R. Roberts. 1985. A critical examination of environmental modelling; modelling the environmental fate of chlorobenzenes using the persistence and fugacity models. Associate Committee on Scientific Criteria for Environmental Quality. National Research Council of Canada. NRCC no. 23990. 206 p.

Asmussen, L.E., A.W. White, E.W. Hauser and J.A. Sheridan. 1977. Movement of 2,4-D in a vegetated waterway. J. Environ. Qual. 6:159-162.

Attaway, H.H., N.D. Camper and M.J.B. Paynter. 1982a. Anaerobic microbial degradation of diuron by pond sediment (herbicides). Pestic. Biochem. Physiol. 17(1):96-101.

Attaway, H.H., M.J.B. Paynter and N.D. Camper. 1982b. Degradation of selected phenylurea herbicides by anaerobic pond sediment. J. Environ. Sci. Health B17(6):683-699.

Bailey, H.C. and D.H.W. Liu. 1980. Lumbriculus variegatus, a benthic oligochaete, as a bioassay organism. American Society for Testing and Materials. ASTM 707. pp. 205-215.

Bailey, G.W., R.R. Swank Jr. and H.P. Nicholson. 1974. Predicting pesticide runoff from agricultural land: a conceptual model. J. Environ. Qual. 3(2):95-102.

Baker, J.L., H.P. Johnson, and J.M. Laflen. 1976. Completion report-effect of tillage systems on runoff losses of pesticides: a simulated rainfall study. Iowa State Water Resour. Res. Inst. Rep. no. ISWRRI-71, Ames, Iowa. 96 p..

Balcomb, R., C.A. Bowen II, D. Wright and M. Law. 1984. Effects on wildlife of at-planting corn applications of carbofuran. J. Wildl. Manage. 48(4):1353-1359.

Ball, I.V., D.S. Gilmer, L.M. Cowardin and J.H. Riechmann. 1975. Survival of wood duck and mallard broods in north-central Minnesota. J. Wildl. Manage. 39:776-780.

Banks, R.C.. 1979. Human related mortality of birds in the United States. U.S. Fish and Wildlife Service. Special Scientific Report. Wildlife No. 215. 16 p.

Barica, J.. 1974. Some observations on internal recycling, regeneration and oscillation of dissolved nitrogen and phosphorus in shallow self-contained lakes. Arch. Hydrobiol. 73:334-360.

Barica, J.. 1975a. Geochemistry and nutrient regime of saline eutrophic lakes in the Erickson-Elphinstone district of southwestern Manitoba. Fisheries and Marine Service Technical Report no. 511. 82 pp.

Barica, J.. 1975b. Summerkill risk in prairie ponds and possibilities of its prediction. J. Fish. Res. Board Can. 32:1283-1288.

Barnett, A.P., E.W. Hauser, A.W. White and J.H. Holladay. 1967. Loss of 2,4-D in washoff from cultivated fallow land. Weeds 15:133-137.

Barnthouse, L.W., D.L. DeAngelis, R.H. Gardner, R.V. O'Neill, C.D. Powers, G.W. Suter II and D.S. Vaughn. 1982. Methodology for environmental risk analysis. Environmental Science Division Publication no. 2023, ORNL/TM-8167. Oak Ridge National Laboratory. Oak Ridge, Tennessee. 66 p.

Barrett, P.R.F.. 1974. The effect of spraying large plots of Nuphar lutea (L.) (yellow water-lily) with glyphosate. Proc. Brit. Weed Contr. Conf. 1:229-232.

Bartell, S.M., R.H. Gardner, R.V. O'Neill and J.M. Giddings. 1983. Error analysis of predicted fate of anthracene in a simulated pond. Environ. Toxicol. Chem. 2:19-28.

Bartonek, J.C. and J.J. Hickey. 1969a. Food habits of canvasbacks, redheads and lesser scaup in Manitoba. Condor 71:280-290.

Bartonek, J.C. and J.J. Hickey. 1969b. Selective feeding by juvenile diving ducks in summer. Auk 86:443-457.

Beaver, D.L.. 1976. Avian populations in herbicide treated brush fields. Auk 93:543-553.

Bednarz, T.. 1981. The effect of 2,4-D acid on green and blue-green algae in unialgal and mixed cultures. Acta Hydrobiol. 23(2):173-182.

Bell, H.L.. 1971. Unpublished data. National Water Quality Laboratory, Duluth, Minnesota.

Bellrose, F.C.. 1976. Ducks, geese and swans of North America. Stackpole Books, Harrisburg, Pennsylvania. 543 p.

Bellrose, F.C.. 1979. Species distribution, habitats and characteristics of breeding dabbling ducks in North America. In Waterfowl and Wetlands an Integrated Review. Proceedings of a symposium held at the 39th Midwest Fish and Wildlife Conference, Madison, Wisconsin.

Bender, M.E. and P.E. Eisele. 1971. Long term effects of pesticides on stream invertebrates. U.S. Nat. Tech. Info. Serv. Rep. no. PB206692.

Bengtson, S.A.. 1971. Variation in the clutch-size in ducks in relation to the food supply. Ibis 113(4):523-526.

Benning, D.S.. 1976. Standard procedures for waterfowl population and habitat surveys. U.S. Fish and Wildlife Service, Office of Migratory Bird Management, Laurel, Maryland.

Bentley, C.F., L.A. Leskiw and M.E. Walmsley. 1983. Major environmental effects, relations and concerns associated with agriculture in Western Canada. Unpublished Report submitted to the "State of the Environment Group". Environment/Statistics Canada contract no. 1656225. 97 p.

Bernard, J.M. and E. Gorham. 1978. Life history aspects of primary production in sedge wetlands. In Freshwater Wetlands: Ecological Processes and Management Potential, R.E. Good, D.F. Whigham and R.L. Simpson (Eds.). Academic Press. pp. 39-52.

Berrie, A.D.. 1976. Detritus, micro-organisms and animals in fresh water. In The Role of Terrestrial and Aquatic Organisms in Decomposition Processes, J.M. Anderson and A. Macfadyen (Eds.). Blackwell Scientific Publications. pp. 323-338.

Best, E.P.H. and J.P. v.d. Wittenboer. 1978. Effects of paraquat on growth and photosynthesis of Ceratophyllum demersum and Elodea canadensis. In Proc. EWRS 5th Symp. on Aquatic Weeds. pp. 157-162.

Best, L.B.. 1972. First-year effects of sagebrush control on two sparrows. J. Wildl. Manage. 36(2):534-544.

Biggs, B.J.F. and T.J. Malthus. 1982. Macroinvertebrates associated with various aquatic macrophytes in the backwaters and lakes of the upper Clutha Valley, New Zealand. New Zealand Journal of Marine and Freshwater Research 16:81-88.

Bird, R.D.. 1961. Ecology of the aspen parkland of western Canada in relation to land use. Canada Dept. of Agriculture. Publication no. 1066. 110 p.

Birk, L.A. and F.E.B. Roadhouse. 1964. Penetration of and persistence in soil of the herbicide atrazine. Can. J. Plant Sci. 44:21-27.

Blackburn, R.D.. 1963. Evaluating herbicides against weeds. Weeds 11:21-24.

Blatt, B.D.. 1976. Reproductive parameters of mallards in relation to age, captivity and geographic origin. Ph.D. Thesis, Michigan State University. Lansing, Michigan. 50 p.

Bluzat, R. and J. Seuge. 1979. Etude de la toxicite chronique de deux insecticides (carbaryl et lindane) à la génération F1 de *Lymnea stagnalis* L. (Mollusque, Gastéropode, Pulmone). I. Croissance des coquilles. Hydrobiologia 65(3):245-255.

Bowden, D.C.. 1974. Review and evaluation of the May waterfowl breeding ground survey. Unpublished Report to the U.S. Fish and Wildlife Service.

Bowman, B.T. and W.W. Sans. 1979. The aqueous solubility of twenty-seven insecticides and related compounds. J. Environ. Sci. Health B14:625-634.

Bowman, B.T. and W.W. Sans. 1982. Adsorption, desorption, soil mobility, aqueous persistence and octanol-water partitioning coefficients of terbufos, terbufos sulfoxide and terbufos sulfone. J. Environ. Sci. Health B17:447-462.

Bowman, B.T. and W.W. Sans. 1983. Further water solubility determinations of insecticidal compounds. J. Environ. Sci. Health B18(2):221-227.

Bowman, M.C., W.L. Oller, T. Cairns, A.B. Gosnell and K.H. Oliver. 1981. Stressed bioassay systems for rapid-screening of pesticide residues. Part I: Evaluation of bioassay systems. Arch. Environ. Contam. Toxicol. 10(1):9-24.

Bowmer, K.H.. 1982a. Adsorption characteristics of seston in irrigation water: implications for the use of aquatic herbicides. Aus. J. Mar. Freshwater Res. 33:443-458.

Bowmer, K.H.. 1982b. Aggregates of particulate matter and aufwuchs on Elodea canadensis in irrigation waters, and inactivation of diquat. Aus. J. Mar. Freshwater Res. 33:589-593.

Boyd, C.E.. 1970. Production, mineral accumulation and pigment concentrations in Typha latifolia and Scirpus americanus. Ecology 51(2):285-290.

Boyd, H.. 1981. Prairie dabbling ducks 1941-1990. Canadian Wildlife Service Progress Note no. 119. Environment Canada. 9 p.

Boyd, H.. 1985. The large-scale impact of agriculture on ducks in the prairie provinces, 1956-1981. Canadian Wildlife Service Progress Note no. 149. Environment Canada. 13 p.

Boyle, T.P.. 1979. Responses of experimental lentic aquatic ecosystems to suppression of rooted macrophytes. In Aquatic Plants, Lake Management, and Ecosystem Consequences of lake Harvesting, J.E. Breck, R.T. Prentki and O.L. Loucks (Eds.). Institute for Environmental Studies, University of Wisconsin-Madison. pp. 269-283.

Boyle, T.P.. 1980. Effects of the aquatic herbicide 2,4-D DMA on the ecology of experimental ponds. Environ. Pollut. 21A:35-49.

Brace, R.K. and F.D. Caswell. 1985. Prairie waterfowl status report: a briefing document. Unpublished report. Canadian Wildlife Service. Western and Northern Region, Environment Canada.

Brace, R.K. and G.W. Pepper. 1984. Impact of agricultural drainage on wildlife. Unpublished Report. Western Provincial Conference on Soil and Water Management. 22 p.

Bradshaw, J.S., E.L. Loveridge, K.P. Rippee, J.L. Peterson, D.A. White, J.R. Barton and D.K. Fuhriman. 1972. Seasonal variation in residues of chlorinated hydrocarbon pesticides in the water of the Utah Lake drainage system, 1970-1971. Pestic. Monit. J. 6(3):166-170.

Bramble, W.C. and W.R. Byrnes. 1972. A long-term ecological study of game food and cover on a sprayed utility right-of-way. Purdue University, Agricultural Experiment Station, Lafayette, Indiana. Research Bulletin no. 885. 20 p.

Breese, M.H. and D.P. Highwood. 1977. Cypermethrin, a new synthetic pyrethroid insecticide. Proceedings of the 1977 British Crop Protection Conference. Pests and Disease 2:641-648.

Brewer, F., T.L. Lavy and R.E. Talbert. 1982. Effects of flooding on dinitroaniline persistence in soybean (Glycine max) and rice (Oryza sativa) rotations. Weed Sci. 30:531-539.

Briggs, G.G.. 1973. A simple relationship between soil sorption of organic chemicals and their octanol/water partition coefficients. Proc. 7th British Insecticide Fungicide Conf. 11:475-478.

Brinson, M.M., A.E. Lugo and S. Brown. 1981. Primary productivity, decomposition and consumer activity in freshwater wetlands. Ann. Rev. Ecol. Syst. 12:123-161.

Brooker, M.P.. 1976. The ecological effects of the use of dalapon and 2,4-D for drainage channel management. I. Flora and chemistry. Arch. Hydrobiol. 78(3):396-412.

Brooker, M.P. and R.W. Edwards. 1973. Effects of the herbicide paraquat on the ecology of a reservoir. I. Botanical and chemical aspects. Freshwater Biol. 3:157-175.

Brooker, M.P. and R.W. Edwards. 1974. Effects of the herbicide paraquat on the ecology of a reservoir. III. Fauna and general discussion. Freshwater Biol. 4:311-335.

Brooker, M.P. and R.W. Edwards. 1975. Aquatic herbicides and the control of water weeds. Water Res. 9:1-15.

Brown, D.S. and E.W. Flagg. 1981. Empirical prediction of organic pollutant sorption in natural sediments. J. Environ. Qual. 10(3):382-386.

Buikema, A.L. Jr., J.G. Geiger and D.R. Lee. 1980. Daphnia toxicity tests. In Aquatic Invertebrate Bioassay, A.L. Buikema Jr. and J. Cairns Jr. (Eds.). American Society for Testing and Materials. ASTM STP 715. pp. 48-69.

Bunting, D.L. and E.B. Robertson Jr. 1975. Lethal and sublethal effects of herbicides on zooplankton species. Research Report no. 43. Water Resources Research Center. The University of Tennessee, Knoxville, Tennessee.

Burns, L.A., D.M. Cline and R.R. Lassiter. 1982. Exposure analysis modeling system (EXAMS). User manual and system documentation. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia. EPA 600/3-82-023.

Burst, H.F.. 1966. A summary of chemical and physical properties of Dursban. Down to Earth 22(3):21-22.

Butler, G.L., T.R. Deason and J.C. O'Kelley. 1975a. Loss of five pesticides from cultures of twenty-one planktonic algae. Bull. Environ. Contam. Toxicol. 13:149-152.

Butler, G.L., T.R. Deason and J.C. O'Kelley. 1975b. The effect of atrazine, 2,4-D, methoxychlor, carbaryl and diazinon on the growth of planktonic algae. British Phycol. J. 10:371-376.

Calderbank, A.. 1970. The fate of paraquat in water. Outlook Agric. 6(3):128-130.

Canadian Agricultural Insect Pest Review. Annual. Research program. Research Branch. Agriculture Canada.

Caro, J.H., H.P. Freeman, D.W. Glotfelty, B.J. Turner and W.M. Edwards. 1973. Dissipation of soil-incorporated carbofuran in the fiel d. J. Agric. Food Chem. 21:1010-1015.

Caro, J.H., H.P. Freeman and B.C. Turner. 1974. Persistence in soil and losses in runoff of soil-incorporated carbaryl in a small watershed. J. Agric. Food Chem. 22:860-863.

Carroll, B.R., G.H. Willis and J.B. Graves. 1981. Permethrin concentrations on cotton plants, persistence in soil and loss in runoff. J. Environ. Qual. 10(4):497-500.

Charnetski, W.A., D. Erickson, G.M. Pullishy and W.E. Kortsch. 1977. Carbofuran degradation in southern Alberta pond and lake water. Environmental Protection Service, Environment Canada. Alberta. 28 p.

Chemagro Division Research Staff. 1974. Guthion^R (azinphosmethyl): organophosphorus insecticide. Residue Rev. 51:123-180.

Chiou, C.T., V.H. Freed, D.W. Schmedding and R.L. Kohnert. 1977. Partition coefficient and bioaccumulation of selected organic chemicals. Environ. Sci. Technol. 11(5):475-478.

Christy, S.L., E.P. Karlander and J.V. Parochetti. 1981. Effects of glyphosate on the growth rate of Chlorella. Weed Sci. 29(1):5-7.

Chura, N.J.. 1961. Food availability and preferences of juvenile mallards. Trans. N. Am. Wildl. Conf. 26:121-134.

Cliath, M.M., W.F. Spencer, W.J. Farmer, T.D. Shoup and R. Grover. 1980. Volatilization of s-ethylN,N-dipropylthio-carbamate from water and wet soil during and after flood irrigation of alfalfa field. J. Agric. Food Chem. 28:610-613.

Coats, J.R., R.L. Metcalf, I.P. Kapoor, L.-C. Chio and P.A. Boyle. 1979. Physical-chemical and biological degradation studies on DDT analogues with altered aliphatic moieties. J. Agric. Food Chem. 27(5):1016-1022.

Coats, J.R. and N.L. O'Donnell-Jeffery. 1979. Toxicity of four synthetic pyrethroid insecticides to rainbow trout. Bull. Environ. Contam. Toxicol. 23:250-255.

Cochrane, D.R., J.D. Pope Jr., H.P. Nicholson and G.W. Bailey. 1967. The persistence of silvex in water and hydrosoil. Water Resour. Res. 3:517-523.

Cochrane, W.P. and J.B. Russell. 1975. Residues in wheat and soil treated with the mixed butyl esters of 2,4-D. Can. J. Plant Sci. 55:323-325.

Collias, N.E. and E.C. Collias. 1963. Selective feeding by wild ducklings of different species. Wilson Bull. 75:6-14.

Conte, F.S. and J.C. Parker. 1975. Effect of aerially-applied malathion on juvenile brown and white shrimp Penaeus aztecus and P. setiferus. Trans. Am. Fish. Soc. 4:793-799.

Cooch, F.G.. 1969. Waterfowl-production habitat requirements. In Saskatoon Wetlands Seminar. Canadian Wildlife Service Report Series no. 6. pp. 5-10.

Cooch, F.G.. 1981. A continental duck population balance for North America. Unpublished Report. Canadian Wildlife Service. Environment Canada.

Coote, D.R.. 1983. Stresses on land under intensive agricultural use. In Stress on Land in Canada, W. Simpson-Lewis, R. Mckechnie and V. Neimanis (Eds.). Lands Directorate Folio no. 6. Environment Canada. pp. 227-257.

Cope, O.B.. 1966. Contamination of the freshwater ecosystem by pesticides. J. Appl. Ecol. (suppl.) 3:33-44.

Cope, O.B., E.M. Wood and G.H. Wallen. 1970. Some chronic effects of 2,4-D on the bluegill (Lepomis macrochirus). Trans. Amer. Fish. Soc. 99(1):1-12.

Correll, D.L. and T.L. Wu. 1982. Atrazine toxicity to submersed vascular plants in simulated estuarine microcosms. Aquat. Bot. 14:151-158.

Corwin, D.L. and W.J. Farmer. 1984. Non single-valued adsorption-desorption of bromacil and diquat by freshwater sediments. Environ. Sci. Technol. 18(7):507-514.

Coulter, M.W. and W.R. Miller. 1968. Nesting biology of black ducks and mallards in northern New England. Vermont Fish and Games Dept. Bull. 68-2. 74 p.

Coupland, R.T.. 1961. A reconsideration of grassland classification in the northern Great Plains of North America. J. Ecol. 49:135-137.

Courtemanch, D.L. and K.E. Gibbs. 1980. Short- and long-term effects of forest spraying of carbaryl (Sevin-4-oil^R) on stream invertebrates. Can. Entomol. 112:271-276.

Cowan, W.F.. 1982. Waterfowl production on zero-tillage farms. Wildl. Soc. Bull. 10:305-308.

Cowardin, L.M. and D.H. Johnson. 1979. Mathematics and mallard management. J. Wildl. Manage. 43(1):18-35.

Cowardin, L.M., D.S. Gilmer and C.W. Shaiffer. 1985. Mallard recruitment in the agricultural environment of North Dakota. Wildl. Monogr. no. 92. J. Wildl. Manage. Suppl. 49(1). 37 p.

Crawford, N.H. and A.S. Donigian Jr.. 1974. Pesticide transport and runoff model for agricultural lands. U.S. Environmental Protection Agency, Athens, Georgia. EPA 600/2-74-013.

Crawford, S.A.. 1981. Successional events (of aquatic vegetation in a farm pond) following simazine application. Hydrobiologia 77(13):217-223.

Crosby, D.G. and R.K. Tucker. 1966. Toxicity of aquatic herbicides to Daphnia magna. Science 154:289-290.

Cross, D.J.. 1966. Approaches toward an assessment of the role of insect food in the ecology of game birds, especially the partridge (Perdix perdix). Ph.D. Thesis, Imperial College, University of London, London, England.

Crossland, N.O.. 1982. Aquatic toxicology of cypermethrin. II. Fate and biological effects in pond experiments. Aquat. Toxicol. 2:205-222.

Crossland, N.O. and K.E. Elgar. 1983. Fate and biological effects of insecticides in ponds. In Pesticide Chemistry: Human Welfare and the Environment. Volume 3. Mode of Action, Metabolism and Toxicology, J. Miyamoto and P.C. Kearney (Eds.). pp. 551-556.

Crossland, N.O. and R.R. Stephenson. 1979. The role of pond studies in assessing the hazard of toxic chemicals to freshwater ecosystems. In Proc. 1979 British Crop Protection Conf. Pests and Diseases. pp. 453-459.

Crossland, N.O., S.W. Shires and D. Bennett. 1982. Aquatic toxicity of cypermethrin. III. Fate and biological effects of spray drift deposits in freshwater adjacent to agricultural land. Aquat. Toxicol. 2:253-270.

Cullimore, D.R.. 1975. The in vitro sensitivity of some species of Chlorophyceae to a selected range of herbicides. Weed Res. 15:401-406.

Cummins, K.W.. 1973. Trophic relations of aquatic insects. Ann. Rev. Entomol. 18:183-206.

Currie, D.J. and J. Kalff. 1984a. A comparison of the abilities of freshwater algae and bacteria to acquire and retain phosphorus. Limnol. Oceanogr. 29(2):298-310.

Currie, D.J. and J. Kalff. 1984b. The relative importance of bacterioplankton and phytoplankton in phosphorus uptake in freshwater. Limnol. Oceanogr. 29(2):311-321.

Currier, W.W., G.B. Maccollom and G.L. Baumann. 1982. Drift residues of air-applied carbaryl in an orchard environment. J. Econ. Entomol. 75:1065-1068.

Daciw, M.. 1982. Herbicides used agriculturally in Western Canada for weed control. Agricultural statistics (annual), Manitoba Agriculture, Winnipeg.

Dad, N.K. and P.S. Tripathi. 1980. Acute toxicity of herbicides to freshwater fish and midge larvae, Chironomus tentans. Environ. Internat. 4:435-437.

Davis, C.B. and A.G. van der Valk. 1978. Litter decomposition in prairie glacial marshes. In Freshwater Wetlands, R.E. Good, D.F. Whigham and R.L. Simpson (Eds.). Academic Press. pp. 99-113.

Davis, E.A. and P.A. Ingebo. 1973. Picloram movement from a chapparal watershed. Water Resour. Res. 9:1304-1313.

Davis, R.B.. 1974. Tubificids alter profiles of redox potential and pH in a profundal lake sediment. Limnol. Oceanogr. 19:342-346.

Day, K.E.. 1986. The acute, chronic and sublethal effects of the synthetic pyrethroid, fenvalerate, on zooplankton in the laboratory and the field. Ph.D. Thesis, University of Guelph.

de Heer, H.. 1979. Measurements and computations on the behavior of the insecticides azinphosmethyl and dimethoate. Agricultural Research Report 884. Centre for Agricultural Publishing and Documentation, Wageningen, The Netherlands.

de la Cruz, A.A.. 1978. Primary production processes: summary and recommendations. In Freshwater Wetlands, R.E. Good, D.F. Whigham and R.L. Simpson (Eds.). Academic Press. pp. 79-86.

Dennis, D.S., W.H. Gillespie, R.A. Maxey and R. Shaw. 1977. Accumulation and persistence of picloram (Tordon 10K) in surface water and bottom sediments in West Virginia. Arch. Environ. Contam. Toxicol. 6:421-433.

Dennis, W.H. Jr., E.P. Meier, W.F. Randall, A.B. Rosencrance and D.H. Rosenblatt. 1979. Degradation of diazinon by sodium hypochlorite. Chemistry and aquatic toxicity. Environ. Sci. Technol. 13(5):594-598.

deNoyelles, F., W.D. Kettle and D.E. Sinn. 1982. The responses of plankton communities in experimental ponds to atrazine, the most heavily used pesticide in the United-States. Ecology 63(5):1285-1293.

Deuel, L.L. Jr., K.W. Brown, F.C. Turner, D.G. Westfall and J.D. Price. 1977. Persistence of propanil, DCA and TCAB in soil and water under flooded rice culture. J. Environ. Qual. 6:127-132.

Deuel, L.E., F.T. Turner, K.W. Brown and J.D. Price. 1978. Persistence and factors affecting dissipation of molinate under flooded rice culture. J. Environ. Qual. 7:373-376.

De Witt, J.R.. 1983. Control of insects. In Proceedings of the Management Alternatives for Biological Farming Workshop, R.B. Dahlgren (Ed.). Iowa Cooperative Wildlife Research Unit. pp. 12-16.

Dirschl, H.J.. 1969. Foods of lesser scaup and blue-winged teal in the Saskatchewan River Delta. J. Wildl. Manage. 33(1):77-87.

Doma, S. and P. Evered. 1977. PP557: Acute toxicity and reproduction studies on first instar and ephippia of Daphnia magna. ICI Plant Protection Division Report no TMJ1455B.

Dortland, R.J.. 1980. Toxicological evaluation of parathion and azinphosmethyl in freshwater model ecosystems. Agric. Res. Rep. (Versl. landbouwk. Onderz.) 898. 112 p.

Doty, H.A., D.L. Trauger and J.R. Serie. 1984. Renesting by canvasback in southwestern Manitoba. J. Wildl. Manage. 48(2):581-584.

Drew, E.A., F.D. Smith and W. Wilkinson. 1977. PP557: ecological effects in three ponds, U.K., 1976-1. Description of ponds and treatment. ICI Plant Protection Division Rep. No. TMJ1473B, July 1977.

Drewien, R.C. and P.F. Springer. 1969. Ecological relationships of breeding blue-winged teal to prairie potholes. In Saskatoon Wetlands Seminar. Canadian Willife Service Report Series no. 6. pp. 102-115.

Driver, E.A.. 1977. Chironomid communities in small prairie ponds: some characteristics and controls. Freshwater Biol. 7:121-133.

Driver, E.A. and D.G. Peden. 1977. The chemistry of surface water in prairie ponds. Hydrobiologia 53(1):33-48.

Driver, E.A., L.G. Sugden and R.J. Kovach. 1974. Calorific, chemical physical values of potential duck foods. Freshwater Biol. 4:281-292.

Duebbert, H.F.. 1969. High nest density and hatching success of ducks on South Dakota CAP land. Trans. N. Am. Wildl. Nat. Resour. Conf. 34:218-228.

Duebbert, H.F. and A.M. Frank. 1984. Value of prairie wetlands to duck broods. Wildl. Soc. Bull. 12:27-34.

Duebbert, H.F. and H.A. Kantrud. 1974. Upland duck nesting related to land use and predator reduction. J. Wildl. Manage. 38:257-265.

Duebbert, H.F. and J.T. Lokemoen. 1976. Duck nesting in fields of undisturbed grass-legume cover. J. Wildl. Manage. 40:39-49.

Duebbert, H.F., J.T. Lokemoen and D.E. Sharp. 1983. Concentrated nesting of mallard and gadwall on Miller Lake Island, North Dakota. J. Wildl. Manage. 47:729-740.

Dvorak, J.. 1970a. Horizontal zonation of macrovegetation, water properties and macrofauna in a littoral stand of Glyceria aquatica (L.) Wahlb. in a pond in South Bohemia. Hydrobiologia 35:17-30.

Dvorak, J.. 1970b. A quantitative study on the macrofauna of stands of emergent vegetation in a carp pond of south-west Bohemia. Rozpr. Cesk. Akad. Ved., Rada Mat. Prir. Ved. 80:63-114.

Dvorak, J.. 1978. Macrofauna of invertebrates in helophyte communities. In Pond Littoral Ecosystems, Structure and Functioning, D. Dykyjová and J. Kvet (Eds.). Ecological Studies no. 28. Springer-Verlag. pp. 389-392.

Dvorak, J. and E.P.H. Best. 1982. Macro-invertebrate communities associated with the macrophytes of Lake Vechten: structural and functional relationships. Hydrobiologia 95:115-126.

Dwernychuk, L.W. and D.A. Boag. 1973. Effect of herbicide-induced changes in vegetation on nesting ducks. Can. Field-Nat. 87:155-165.

Dwyer, T.J.. 1970. Waterfowl breeding habitat in agricultural and non-agricultural land in Manitoba. J. Wildl. Manage. 34(1):130-136.

Dwyer, T.J., G.L. Krapu and D.M. Janke. 1979. Use of prairie pothole habitat by breeding mallards. J. Wildl. Manage. 43(2):526-531.

Dzubin, A.. 1969a. Comments on carrying capacity of small ponds for ducks and possible effects of density on mallard production. In Saskatoon Wetlands Seminar. Canadian Wildlife Service Report Series no. 6. pp. 138-168.

Dzubin, A.. 1969b. Assessing breeding populations of ducks by ground counts. In Saskatoon Wetlands Seminar. Canadian Wildlife Service Report Series no. 6. pp. 178-230.

Dzubin, A. and J.B. Gollop. 1972. Aspects of mallard breeding ecology in Canadian parkland and grassland. In Population Ecology of Migratory Birds. U.S. Fish and Wildlife Service Res. Rep. no. 2. pp. 113-152.

Edwards, M.J., E.A. Drew, J.P. Ford and R.J. Ward. 1977. PP557: ecological effects in three ponds, U.K., 1976-2. Residues in water. ICI Plant Protection Division Rep. No. TMJ1474B, July 1977.

Edwards, W.M.. 1972. Agricultural chemical pollution as affected by reduced tillage systems. In Proc. No-tillage Systems Symp., 21 Feb. 1972, Ohio State Univ., Columbus. pp. 30-40.

Edwards, W.M. and B.L. Glass. 1971. Methoxychlor and 2,4,5-T in lysimeter percolation and runoff water. Bull. Environ. Contam. Toxicol. 6:81-84.

Edwards, W.M., G.B. Triplett Jr. and R.M. Kramer. 1980. A watershed study of glyphosate transport in runoff. J. Environ. Qual. 9(4):661-665.

Ehrlich, P.R., A.H. Ehrlich and J.P. Holdren. 1977. Ecoscience: population, resources, environment. W.H. Freeman and Co.. San Francisco. 1051 p.

Eisenlohr, Wm.S. Jr..1966. Water loss from a natural pond through transpiration by hydrophytes. Water Resour. Res. 2:443-453.

Eisenlohr, Wm.S. Jr.. 1969. Hydrology of small water areas in the prairie pothole region. In Saskatoon Wetlands Seminar. Canadian Wildlife Service Report Series no. 6. pp. 35-39.

Eisler, R.. 1970. Factors affecting pesticide induced toxicity to estuarine fish. Tech. Paper Fish Wildl. Ser. no. 15.

El-Dib, M.A. and O.A. Aly. 1976a. Persistence of some phenylamide pesticides in the aquatic environment. I. Hydrolysis. Water Res. 10(12):1047-1050.

El-Dib, M.A. and O.A. Aly. 1976b. Persistence of some phenylamide pesticides in the aquatic environment. II. Adsorption on clay minerals. Water Res. 10(12):1051-1053.

El-Dib, M.A. and O.A. Aly. 1976c. Persistence of some phenylamide pesticides in the aquatic environment. III. Biological degredation. Water Res. 10(12):1055-1059.

Ellgehausen, H., J.A. Guth and H.O. Esser. 1980. Factors determining the bioaccumulation potential of pesticides in the individual compartments of aquatic food chains. Ecotoxicol. Environ. Saf. 4:134-157.

Elliott, M. and N.F. Janes. 1978. Synthetic pyrethroids: a new class of insecticide. Chem Soc. Rev. 7:473.

Elliott, M., N.F. Janes and C. Potter. 1978. The future of pyrethroids in insect control. Ann. Rev. Entomol. 23: 443-469.

Emmerich, W.E., J.D. Helmer, K.G. Renard and L.J. Lane. Fate and effectiveness of tebuthiuron applied to a rangeland watershed. 1984. J. Environ. Qual. 13(3):382-386.

Energy, Mines and Resources. 1974. The national atlas of Canada. Macmillan Company of Canada Limited in association with the Dept. of Energy, Mines and Resources and Information Canada. 254 p.

Evans, D.M.. 1978. Aquatic weed control with the isopropylamine salt of N-phosphonomethyl glycine. In Proc. EWRC 5th Symp. Aquatic Weeds. pp. 171-178.

Evans, R.D. and C.W. Wolfe. 1967. Waterfowl production in the rainwater basin area of Nebraska. J. Wildl. Manage. 31(4):788-794.

Everts, J.W., K. van Frankenhuyzen, B. Roman and J.H. Koeman. 1983. Side-effects of experimental pyrethroid applications for the control of tse-tse flies in a riverine forest habitat (Africa). Arch. Environ. Contam. Toxicol. 12:91-97.

Ewen, A.B. and M.K. Mukerji. 1985. Another bait for grasshopper control. Biweekly Letter no. 118. Agriculture Research Station. Saskatoon, Saskatchewan.

FAO/WHO. 1980. Pesticide residues in food, 1979: report of the joint meeting of the FAO panel of experts on pesticide residues in food and the environment and the WHO expert group on pesticide residues. Rome. Food and Agriculture Organization of the United Nations. 90 p.

Federle, P.F. and W.J. Collins. 1976. Insecticide toxicity to three insects from Ohio ponds. Ohio J. Sci. 76(1):19-24.

Filion, D.B.. 1967. The abundance and distribution of benthic fauna of three mountain reservoirs on the Kananaskis river in Alberta. J. Appl. Ecol. 4:1-11.

Fitzmayer, K.M., J.G. Geiger and M.J. Van den Avyle. 1982. Effects of chronic exposure to simazine on the cladoceran, Daphnia pulex. Arch. Environ. Contam. Toxicol. 11:603-609.

Flickinger, E.L., K.A. King, W.F. Stoutard and M.M. Mohn. 1980. Wildlife hazards from furadan 3G applications to rice in Texas. J. Wildl. Manage. 44(1):190-197.

Folmar, L.C.. 1978. Avoidance chamber responses of mayfly nymphs exposed to eight herbicides. Bull. Environ. Contam. Toxicol. 19(3):312-318.

Folmar, L.C., H.O. Sanders and A.M. Julin. 1979. Toxicity of the herbicide glyphosate and several of its formulations to fish and aquatic invertebrates. Arch. Environ. Contam. Toxicol. 8:269-278.

Forney, D.R. and D.E. Davis. 1981. Effects of low concentrations of herbicides on submerged aquatic plants. Weed Sci. 29:677-685.

Foy, C.L.. 1975. The chlorinated alyphatic acids. *In* Herbicides, Chemistry, Degradation and Mode of Action, P.C. Kearney and D.D. Kaufman (Eds.). Marcel Dekker Inc., New York, Basel. pp. 399-452.

Frank, P.A., R.J. Demint and R.D. Comes. 1970. Herbicides in irrigation water following canal-bank treatment for weed control. Weed Sci. 18(6):687-692.

Frank, R. and G.J. Sirons. 1979. Atrazine: its use in corn production and its loss to stream waters in southern Ontario, 1975-1977. Sci. Total Environ. 12:223-239.

Frear, D.E.H. and J.E. Boyd. 1967. Use of *Daphnia magna* for the microbioassay of pesticides. I. Development of standardized techniques for rearing *Daphnia* and preparation of dosage-mortality curves for pesticides. J. Econ. Entomol. 60:1228-1236.

Fredeen, F.J.H., I.G. Saha and M.H. Balba. 1975. Residues of methoxychlor and other chlorinated hydrocarbons in water, sand and selected fauna following injections of methoxychlor blackfly larvicide into the Saskatchewan River, 1972. Pestic. Monit. J. 8(4):241-246.

Freed, V.H., D. Schmedding, R. Kohnert and R. Haque. 1979. Physical chemical properties of several organophosphates: some implication in environmental and biological behavior. Pestic. Biochem. Physiol. 10:203-211.

Friesen, M.K., T.D. Galloway and J.F. Flannagan. 1983. Toxicity of the insecticide permethrin in water and sediment to nymphs of the burrowing mayfly *Hexagenia rigida* (Ephemeroptera:ephemeridae). Can. Entomol. 115:1007-1014.

FWPCA. 1968. Water quality criteria. Report of the National Tech. Adm. Comm. to Secr. of the Interior Fed. Water Pollution Control Administration. U.S. Dept. of the Interior.

Gaaboub, I.A. and T.A. Abu-Hashish. 1981. Susceptibility of egyptian *Culex pipiens* L. to six synthetic pyrethroids. Insect Sci. Appl. 1(3):297-301.

Gaaboub, I.A., I.A. Rawash, A. Tag-El-Din and M.A.T. Hassanein. 1981. Joint action of six herbicides with malathion against mosquito larvae of Culex pipiens L.. Toxicology 20:61-70.

Gage, S.H. and M.K. Mukerji. 1977. A perspective of grasshopper population distribution in Saskatechewan and interrelationship with weather. Environ. Entomol. 6(3):469-479.

Gage, S.H. and M.K. Mukerji. 1978. Crop losses associated with grasshoppers in relation to economics of crop protection. J. Econ. Entomol. 71(3):487-498.

Gates, J.M.. 1962. Breeding biology of the gadwall in northern Utah. Wilson Bull. 74:43-67.

Geller, A.. 1980. Studies on the degradation of atrazine by bacterial communities enriched from various biotopes. Arch. Environ. Contam. Toxicol. 9:289-305.

George, J.P., H.G. Hingorani and K.S. Rao. 1982. Herbicide toxicity to fish-food organisms. Environ. Pollut. A28:183-188.

Gerking, S.D.. 1962. Production and food utilization in a population of bluegill sunfish. Ecol. Monogr. 32:31-87.

Gerrish, N. and J.M. Bristow. 1979. Macroinvertebrate associations with aquatic macrophytes and artificial substrates. J. Great Lakes Res. 5(1):69-72.

Gersich, F.M., D.L. Hopkins and D.P. Milazzo. 1985. Acute and chronic toxicity of technical picloram (4-amino-3,5,6-trichloropicolinic acid) to Daphnia magna Straus. Bull. Environ. Contam. Toxicol. 35:121-126.

Getsinger, K.D., G.J. Davis and M.M. Brinson. 1982. Changes in a Myriophyllum spicatum L. community following 2,4-D treatment. J. Aquat. Plant Manage. 20:4-8.

Getty, C. and W. Wilkinson. 1978. PP557: ecological effects on three ponds, U.K., 1976-77. Observations on invertebrates and amphibia up to 1 year after treatment. ICI Plant Protection Division Report No. RJ0041B November 1978.

Ghassemi, M., L. Fargo, P. Painter, S. Quinlivan, R. Scofield and A. Takata. 1981. Environmental fates and impacts of major forest use pesticides. TRW, Redondo Beach, California.

Gibbs, K.E., T.M. Mingo and D.L. Courtemanch. 1984. Persistence of carbaryl (Sevin-4-oil) in woodland ponds and its effects on pond macroinvertebrates following forest spraying. Can. Entomol. 116:203-213.

Gilderhus, P.A.. 1967. Effects of diquat on bluegills and their food organisms. The Progressive Fish-Culturist, April 1967.

Gile, J.D.. 1983. Relative airborne losses of commercial 2,4-D formulations from a simulated wheat field. Arch. Environ. Contam. Toxicol. 12:465-469.

Gillis, C.. 1984. The farmer as an operator. In Proc. Symp. the Future Role of Aviation in Agriculture, P.W. Voisey (Ed.). Associate Committee on Agricultural and Forestry Aviation. AFA-TN-17, NRC no. 23504. pp. 249-258.

Girard, G.L.. 1939. Notes on the life history of the shoveler. Trans. N. Am. Wildl. Conf. 4:364-371.

Girman, G.R.. 1976. The effects of a number of herbicides upon photosynthesis and heterotrophy of naturally occurring algal and bacterial communities in Delta Marsh, Manitoba. M.Sc. Thesis, University of Manitoba.

Göhlich, H.. 1983. Formation of drift and basic considerations in its reduction. Int. Union Pure Appl. Chem. 4:271-280.

Goldsborough, L.G. and G.G.C. Robinson. 1983. The effect of two triazine herbicides on the productivity of freshwater marsh periphyton. Aquat. Toxicol. 4:95-112.

Gollop, J.B.. 1965. Wetland inventories in Western Canada. In International Union of Game Biologists, Transactions of the 6th congress, Oct. 7-12, 1963. The Nature Conservancy, London. pp. 249-264.

Gomaa, H.M., I.H. Suffet and S.D. Faust. 1969. Kinetics of hydrolysis of diazinon and diazoxon. Residue Rev. 29:171-190.

Goodman, D.C. and H.I. Fisher. 1962. Functional anatomy of the feeding apparatus in waterfowl (Aves:Anatidae). Southern Illinois University Press. Carbondale, Illinois.

Gorden, R.W., S.W. Waite, P. Tazik and M.J. Wiley. 1982. The effect of simazine and endothall in aquatic macrophyte control. In 6th International Symposium on Aquatic Weeds. EWRS, Novi Sad. pp. 287-296.

Grau, C.R.. 1968. Avian embryo nutrition. Fed. Proc. 27(1):185-192.

Gray, H.E.. 1965. Dursban, a new organophosphorus insecticide. Down to Earth 21(3):2,26,27.

Griffith, R.. 1948. Improving waterfowl habitat. Trans. N. Am. Wildl. Conf. 13:609-618.

Grover, R., J. Maybank and K. Yoshida. 1972. Droplet and vapour drift from butylester and dimethylamine salt of 2,4-D. Weed Sci. 20:320-324.

Grover, R. and A.E. Smith. 1974. Adsorption studies with the acid and dimethylamine forms of 2,4-D and dicamba. Can. J. Soil Sci. 54:179-186.

Grover, R., S.R. Shewchuk, A.J. Cessna, A.E. Smith and J.H. Hunter. 1985. Fate of 2,4-D iso-octyl ester after application to a wheat field. J. Environ. Qual. 14(2):203-210.

Grue, C.E., L.R. De Weese, P. Mineau, G.A. Swanson, J.R. Foster, P.M. Arnold, J.N. Huckins, P.J. Sheehan, W.K. Marshall and A.P. Ludden. 1986. Potential impacts of agricultural chemicals on waterfowl and other wildlife inhabiting prairie wetlands: an evaluation of research needs and approaches. Trans. 51st N. Am. Wildl. Nat. Res. Conf.. (In press).

Grue, C.E., W.J. Fleming, D.G. Busby and E.F. Hill. 1983. Assessing hazards of organophosphate pesticides to wildlife. Trans. N. Am. Wildl. Nat. Res. Conf. 48:200-220.

Grzenda, A.R., H.P. Nicholson and W.S. Cox. 1966. Persistence of four herbicides in pond water. J. Amer. Water Works Assoc. 58:326-332.

Gummer, Wm.D.. 1979. Pesticide residues in prairie surface waters. Inland Waters Directorate. Western and Northern Region. Environment Canada.

Haines, T.A.. 1981. Effect of an aerial application of carbaryl on brook trout (Salvelinus fontinalis). Bull. Environ. Contam. Toxicol. 27:534-542.

Hall, D.J., W.E. Cooper and E.E. Werner. 1970. An experimental approach to the production dynamics and structure of freshwater animal communities. Limnol. Oceanogr. 15:839-928.

Hall, J.K.. 1974. Erosional losses of s-triazine herbicides. J. Environ. Qual. 3:174-180.

Hall, J.K., M. Pawlus and E.R. Higgins. 1972. Losses of atrazine in runoff water and soil sediment. J. Environ. Qual. 1:172-176.

Hamaker, J.W. and J.M. Thompson. 1972. Adsorption. In Organic Chemicals in the Soil Environment, C.A.I. Goring and J.W. Hamaker (Eds.). Marcel Decker Inc., New York. pp. 49-143.

Hance, R.J.. 1965. The adsorption of urea and some of its derivatives by a variety of soils. Weed Res. 5:98-107.

Hanson, W.R.. 1952. Effects of some herbicides and insecticides on biota of North Dakota marshes. J. Wildl. Manage. 16(3):299-308.

Hardy, J.L.. 1966. Effect of Tordon herbicides on aquatic chain organisms. Down to Earth 22(2):11-13.

Hartland-Rowe, R.. 1966. The fauna and ecology of temporary pools in Western Canada. Verh. Int. Verein. Limnol. 16:577-584.

Hashimoto, Y. and Y. Nishiuchi. 1981. Establishment of bioassay methods for the evaluation of acute toxicity of pesticides to aquatic organisms. J. Pestic. Sci. 6(2):257-263.

Hawxby, K.W. and R. Mehta. 1979. The fate of aquazine in a small pond. Proc. Okla. Acad. Sci. 59:16-19.

Hay, J.R.. 1980. Pesticides and herbicides: present and anticipated use by 1990. Canada West Foundation Publ. no. 80-131. Calgary. 15 p.

Hayden, B.J. and A.E. Smith. 1980. Comparison of the persistence of ethalfluralin and trifluralin in Saskatchewan field soils. Bull. Environ. Contam. Toxicol. 25:508-511.

Heckman, C.W.. 1981. Long-term effects of intensive pesticide applications on the aquatic community in orchard drainage ditches near Hamburg, Germany. Arch. Environ. Contam. Toxicol. 10:393-426.

Hedlund, R.T. and C.R. Youngson. 1972. The rates of photodecomposition of picloram in aquaeous systems. In Fate of Organic Pesticides in the Aquatic Environment, S.D. Faust (Ed.). Advances in Chemistry Series 111. pp. 159-172.

Helling, C.S.. 1971. Pesticide mobility in soils: application of soil thin-layer chromatography. Soil Sci. Soc. Amer. Proc. 35:737-743.

Helling, C.S.. 1976. Dinitroaniline herbicides in soils. J. Environ. Qual. 5:1-15.

Hemmett, R.B. Jr. and S.D. Faust. 1969. Biodegradation kinetic of 2,4-dichlorophenoxy acetic acid by aquatic organisms. Residue Rev. 29:191-207.

Higgins, K.F.. 1977. Duck nesting in intensively farmed areas of North Dakota. J. Wildl. Manage. 41(2):232-242.

Hill, B.D.. 1981. Persistence and distribution of fenvalerate residues in soil under field and laboratory conditions. J. Agric. Food Chem. 29:107-110.

Hill, B.D. and S. Kinniburgh. 1984. Aerial deposition of the synthetic pyrethroid deltamethrin. Proc. Symp. on the Future Role of Aviation in Agriculture, P.W. Voisey (Ed.). The National Research Council of Canada Associate Committee on Agricultural and Forestry Aviation. NRC no. 23504. AFA-TN-17.

Hill, E.F. and W.J. Fleming. 1982. Anticholinesterase poisoning of birds: field monitoring and diagnosis of acute poisoning. Environ. Toxicol. Chem. 1:27-38.

Hill, E.F., R.G. Heath, J.W. Spann and J.D. Williams. 1975. Lethal dietary toxicities of environmental pollutants to birds. U.S. Fish and Wildlife Service Special Scientific Report, Wildlife no. 191. 61 p.

Hines, J.E. and G.J. Mitchell. 1983. Gadwall nest-site selection and nesting success. J. Wildl. Manage. 47:1063-1071.

Hilsenhoff, W.. 1966. Effect of diquat on aquatic insects and related animals. J. Econ. Entomol. 59(6):1520-1521.

Hirano, C.. 1981. Persistence of diazinon in water of a paddy field and a stream receiving paddy field water. Nippon Oyo Dobutsu Konchu Gakkaishi 25(3):191-193.

Hobbie, J.E., T. Traaen, P. Rublee, J.P. Reed, M.C. McMiller and T. Fenchel. 1980. Decomposers, bacteria and microbenthos. In Limnology of tundra ponds, J.E. Hobbie (Ed.). US/IBP Synth. Ser. 13. pp. 340-387.

Hochbaum, H.A.. 1944. The canvasback on a prairie marsh. The American Wildlife Institute. 201 p.

Hodson, P.V., C.N. Ross, A.J. Niimi and D.J. Spry. 1977. Statistical considerations in planning aquatic bioassays. In Proc. of the 3rd Aquatic Toxicity Workshop, W.R. Parker, E. Pessah, P.G. Wells and J.F. Westlake (Eds.). EPS Report EPS-5-AR-77-1. Environment Canada.

Hoffman, D.J. and P.H. Albers. 1984. Evaluation of potential embryotoxicity and teratogenicity of 42 herbicides, insecticides and petroleum contaminants to mallard eggs. Arch. Environ. Contam. Toxicol. 13(1):15-27.

Hooper, N.M. and G.G.C. Robinson. 1976. Primary production of epiphytic algae in a marsh pond. Can. J. Bot. 54:2810-2815.

Hooper-Reid, N.M. and G.G.C. Robinson. 1978a. Seasonal dynamics of epiphytic algal growth in a marsh pond: productivity, standing crop, and community composition. Can. J. Bot. 56:2434-2440.

Hooper-Reid, N.M. and G.G.C. Robinson. 1978b. Seasonal dynamics of epiphytic algal growth in a marsh pond: composition, metabolism, and nutrient availability. Can. J. Bot. 56:2441-2448.

Hudson, R.H., R.K. Tucker and M.A. Haegele. 1984. Handbook of toxicity of pesticides to wildlife. Second edition. U.S. Fish and Wildlife Service Publication 153. 90 p.

Hughes, D.N., M.G. Boyer, M.H. Papst, C.D. Fowle, G.A.V. Rees and P. Baulu. 1980. Persistence of three organophosphorus insecticides in artificial ponds and some biological implications. Arch. Environ. Contam. Toxicol. 9:269-279.

Hunter, J.H. and E.H. Stobbe. 1972. Movement and persistence of picloram in soil. Weed Sci. 20:486-489.

Hunter, M.L. Jr., J.W. Witham and H. Dow. 1984. Effects of carbaryl-induced depression in invertebrate abundance on the growth and behavior of American black duck and mallard ducklings. Can. J. Zool. 62:452-456.

Hurlbert, S.H.. 1975. Secondary effects of pesticides on aquatic ecosystems. Residue Rev. 57:81-148.

Hurlbert, S.H., M.S. Mulla, J.O. Keith, W.E. Westlake and M.E. Düsch. 1970. Biological effects and persistence of Dursban in freshwater ponds. J. Econ. Entomol. 63(1):43-52.

Hurlbert, S.H., M.S. Mulla and H.R. Wilson. 1972. Effects of an organophosphorus insecticide on the phytoplankton, zooplankton, and insect populations of freshwater ponds. Ecol. Monogr. 42:269-299.

Isensee, A.R., P.C. Kearney and G.E. Jones. 1979. Modeling aquatic ecosystems for metabolic studies. In Pesticide and Xenobiotic Metabolism in Aquatic Organisms, M.A.Q. Khan, J.J. Lech and J.J. Menn (Eds.). ACS Symposium Series, American Chemical Society, Washington. pp. 195-216.

Jensen, K.I.N. and E.R. Kimball. 1980. Persistence of dinitramine and trifluralin in Nova Scotia, Canada. Bull. Environ. Contam. Toxicol. 24:238-243.

Jervis, R.A.. 1969. Primary production in the freshwater marsh ecosystem of Troy Meadows, New Jersey. Bull. Torrey Bot. Club 96(2):209-231.

Jeyasingam, D.N.T., B. Thayumanavan and S. Krishnasnamy. 1978. The relative toxicities of insecticides on aquatic insect Eretes stietieus (Linn.) (Coleoptera:Dytiscidae). J. Madura Univ. 7(1):85-87.

Johnson, W.W. and M.T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Summaries of toxicity tests conducted at Columbia National Fisheries Research Laboratory, 1965-1978. U.S. Fish and Wildlife Service Resource Publication 137.

Jolly, A.L. Jr., J.W. Avault Jr., K.L. Koonce and J.B. Graves. 1978. Acute toxicity of permethrin to several aquatic animals. Trans. Am. Fish. Soc. 107(6):825-827.

Jones, T.W. and P.S. Estes. 1984. Uptake and phytotoxicity of soil-sorbed atrazine for the submerged aquatic plant, Potamogeton perfoliatus L. Arch. Environ. Contam. Toxicol. 13:237-241.

Jones, T.W. and L. Winchell. 1984. Uptake and photosynthetic inhibition by atrazine and its degradation products on four species of submerged vascular plants. J. Environ. Qual. 13(2):243-247.

Joyner, D.E.. 1980. Influence of invertebrates on pond selection by ducks in Ontario. J. Wildl. Manage. 44(3):700-705.

Joyner, D.E.. 1982. Abundance and availability of invertebrates in ponds in relation to dietary requirements of mallards and blue-winged teal at Luther Marsh. Ont. Field Biol. 36(1):19-34.

Julin, A.M. and F.J. Gramlich. 1978. Effects of orthene on songbird brain cholinesterase in Maine 1977. In Environmental Monitoring of Cooperative Spruce Budworm Projects, Maine 1976 and 1977. Maine Department of Conservation, Bureau of Forestry, Augusta, Maine, U.S.A.. pp. 183-187.

Kadlec, J.A.. 1962. Effects of a drawdown on a waterfowl impoundment. Ecology 43:267-281.

Kadlec, J.A.. 1979. Nitrogen and phosphorus dynamics in inland freshwater wetlands. In Waterfowl and Wetlands. An Integrated Review, T.A. Bookhout (Ed.). Proc. Symp. 39th Midwest Fish and Wildlife Conf., Madison, Wisconsin, 1977. pp. 17-41.

Kaminski, R.M. and H.H. Prince. 1984. Dabbling duck-habitat associations during spring in Delta Marsh, Manitoba. J. Wildl. Manage. 48(1):37-50.

Kanazawa, J. 1981. Measurements of the bioconcentration factors of pesticides by freshwater fish and their correlation with physiochemical properties or acute toxicities. Pestic. Sci. 12:417-424.

Kantrud, H.A. and R.E. Stewart. 1977. Use of natural basin wetlands by breeding waterfowl in North Dakota. J. Wildl. Manage. 41(2):243-253.

Kantrud, H.A. and R.E. Stewart. 1984. Ecological distribution and crude density of breeding birds on prairie wetlands. J. Wildl. Manage. 48(2):426-437.

Kapoor, I.P., R.L. Metcalf, F.R. Nystrom and G.K. Sangha. 1970. Comparative metabolism of methoxychlor, methiochlor and DDT in mouse, insects and in model ecosystem. J. Agric. Food Chem. 18:1145-1152.

Kapoor, I., R.L. Metcalf, A.S. Hirwe, J.R. Coats and M.S. Khalsa. 1973. Structure activity correlations of biodegradability of DDT analogs. J. Agric. Food Chem. 21(2):310-315.

Karickhoff, S.W. and D.S. Brown. 1978. Paraquat sorption as a function of particle size in natural sediments. J. Environ. Qual. 7:246-252.

Karickhoff, S.W., D.S. Brown and T.A. Scott. 1979. Sorption of hydrophobic pollutants on natural sediments. Water Res. 13:241-248.

Kaufman, D.D. and P.C. Kearney. 1970. Microbial degradation of triazine herbicides. Residue Rev. 32:235-265.

Kaufman, D.D., B.A. Russel, C.S. Helling and A.J. Kayser. 1981. Movement of cypermethrin, decamethrin and permethrin and their degradation products in soil. J. Agric. Food Chem. 29:239-245.

Kaupke, C.R. and W.E. Yates. 1966. Physical properties and drift characteristics of viscosity-modified agricultural sprays. Trans. ASAE 9(6):797-799.

Kaushik, N.K., G.L. Stephenson, K.R. Solomon and K.E. Day. 1985. Evaluation of the impacts of permethrin on zooplankton communities using limnocorrals. II. Impact on zooplankton. Can. J. Fish. Aquat. Sci. 42:77-85.

Keast, A.. 1984. The introduced aquatic macrophyte Myriophyllum spicatum as habitat for fish and their invertebrate prey. Can. J. Zool. 62:1289-1303.

Keith, L.B.. 1961. A study of waterfowl ecology on small impoundments in southeastern Alberta. Wildl. Monogr. 6. 88 p.

Kells, J.J., C.E. Reick, R.L. Blevins and W.M. Muir. 1980. Atrazine dissipation as affected by surface pH and tillage. Weed Sci. 28(1):101-104.

Kenaga, E.E.. 1974. Toxicological and residue data useful in the environmental safety evaluation of dalapon. Residue Rev. 53:109-151.

Kenaga, E.E.. 1978. Test organisms and methods useful for early assessment of acute toxicity of chemicals. Environ. Sci. Technol. 12(12):1322-1329.

Kenaga, E.E.. 1980. Correlation of bioconcentration factors of chemicals in aquatic and terrestrial organisms with their physical and chemical properties. Environ. Sci. Technol. 14(5):553-556.

Kenaga, E.E.. 1982a. The use of environmental toxicology and chemistry data in hazard assessment: progress, needs, challenges. Environ. Toxicol. Chem. 1:69-79.

Kenaga, E.E.. 1982b. Predictability of chronic toxicity of chemicals in fish and aquatic invertebrates. Environ. Toxicol. Chem. 1:347-358.

Kenaga, E.E., W.K. Whiney, J.L. Kardy and A.E. Doty. 1965. Laboratory tests with Dursban insecticide. J. Econ. Entomol. 58(6):1043-1050.

Kennedy, H.D. and D.F. Walsh. 1970. Effects of malathion on two warmwater fishes and aquatic invertebrates in ponds. U.S. Fish and Wildlife Service. Technical Paper of the Bureau of Sport Fisheries and Wildlife no. 55.

Kennedy, H.D., L.L. Eller and D.F. Walsh. 1970. Chronic effects of methoxychlor on bluegills and aquatic invertebrates. U.S. Fish and Wildlife Service. Technical Paper of the Bureau of Sport Fisheries and Wildlife no. 53.

Keys, C.H. and H.A. Friesen. 1968. Persistence of picloram activity in soil. Weed Sci. 16:341-343.

Khan, S.U.. 1974. Adsorption of 2,4-D from aquaeous solution by fulvic acid-clay complex. Environ. Sci. Technol. 8:236-238.

Khan, S.U., A. Belanger, E.J. Hogue, H.A. Hamilton and S.P. Mathur. 1976. Residues of paraquat and linuron in an organic soil and their uptake by onions, lettuce and carrots. Can. J. Soil Sci. 56:407-412.

Khan, S.U., P.B. Marriage and A.S. Hamill. 1981. Effects of atrazine treatment of a corn field using different application methods, times, and additives on the persistence of residues in soil and their uptake by oat plants. J. Agric. Food Chem. 29:216-219.

King, G.J.. 1978. Rural aircraft utilization in Canada 1976. Agriculture Canada publication no. 78/11. 71 p.

Kingsbury, P.D.. 1976. Studies of the impact of aerial applications of the synthetic pyrethroid NRDC-143 on aquatic ecosystems. Chemical Control Research Institute Information Report CC-X-127. 111 p.

Kingsbury, P.D. and D.P. Kreutzweiser. 1979. Impact of double applications of permethrin on forest streams and ponds. Forest Pest Management Institute. Report FPM-X-27. Canadian Forestry Service. 42 p.

Kingsbury, P.D. and D.P. Kreutzweiser. 1980. Dosage-effect studies on the impact of permethrin on trout streams. Forest Pest Management Institute. Report FPM-X-31. Canadian Forestry Service. 57 p.

Kirsch, L.M.. 1969. Waterfowl production in relation to grazing. J. Wildl. Manage. 33(4):821-828.

Klaassen, H.E. and A.M. Kadoum. 1979. Distribution and retention of atrazine and carbofuran in farm pond ecosystems. Arch. Environ. Contam. Toxicol. 8:345-353.

Kling, H.. 1975. Phytoplankton successions and species distribution in prairie ponds of the Erickson-Elphinstone district, southwestern Manitoba. Fisheries and Marine Service Technical Report no. 512. 31 p.

Klopatek, J.M.. 1975. The role of emergent macrophytes in mineral cycling in a freshwater marsh. In Mineral Cycling in Southeastern Ecosystems, F.G. Howell, J.B. Gentry and M.H. Smith. Energy Research and Development Administration. pp. 367-393.

Koren, E., C.L. Foy and F.M. Ashton. 1969. Adsorption, volatility and migration of thiocarbamate herbicides in soils. Weed Sci. 17:148-153.

Knauf, W. and E.F. Schulze. 1972. Langzeiteinfluss subletaler herbiziddosen auf einige vertreter der wasserfauna and flora am beispiel linuron und monolinuron. Schiftenr. ver Wasser-Boden-Lufthyg. Berlin-Dahlem 37:231.

Korinkova, J.. 1971. Quantitative relations between submerged macrophytes and populations of invertebrates in a carp pond. Hidrobiologia 12:377-382.

Kosinski, R.J. and M.G. Merkle. 1984. The effect of four terrestrial herbicides on the productivity of artificial stream algal communities. J. Environ. Qual. 13(1):75-82.

Krapu, G.L.. 1974a. Feeding ecology of pintail hens during reproduction. Auk 91:278-290.

Krapu, G.L.. 1974b. Foods of breeding pintails in North Dakota. J. Wildl. Manage. 38(3):408-417.

Krapu, G.L.. 1979. Nutrition of female dabbling ducks during reproduction. In Waterfowl and Wetlands. An Integrated Review, T.A. Bookhout (ed.). Proc. Symp. 39th Midwest Fish and Wildlife Conf., Madison, Wisconsin, 1977. pp. 59-69.

Krapu, G.L., A.T. Klett and D.G. Jorde. 1983. The effect of variable spring water conditions on mallard reproduction. Auk 100:689-698.

Krapu, G.L. and G.A. Swanson. 1975. Some nutritional aspects of reproduction in prairie nesting pintails. J. Wildl. Manage. 39(1):156-162.

Krapu, G.L., L.G. Talent and T.J. Dwyer. 1979. Marsh nesting by mallards. Wildl. Soc. Bull. 7(2):104-110.

Kratky, B.A. and G.F. Warren. 1971. The use of three simple, rapid bioassays on forty-two herbicides. Weed Res. 11:257-262.

Kreutzweiser, D.P. and P.D. Kingsbury. 1982. Recovery of stream benthos and its utilization by native fish following high dosage permethrin applications. Forest Pest Management Institute Information Report FPM-X-59. Canadian Forestry Service. 18 p.

Kroeger, P.G. and G.G. Shaw. 1982. Effect of aerial application of picloram on vegetation, birds and small mammals in Buck Coulee, Porcupine Hills, Alberta. Canadian Wildlife Service. Interim Report. 60 p.

Krull, J.N.. 1970. Aquatic plant-macroinvertebrate associations and waterfowl. J. Wildl. Manage. 34(4):707-718.

Kuhr, R.J. and H.W. Dorough. 1976. Carbamate insecticides: chemistry, biochemistry and toxicology. CRC Press, Cleveland. 301 p.

Laing, H.E.. 1941. Effect of concentration of oxygen and pressure of water upon growth of rhyzomes of semi-submerged water plants. Bot. Gaz. 102:712-724.

Lakota, S., A. Raszka and I. Kupczak. 1981. Toxic effects of cartap, carbaryl and propoxur on some aquatic organisms. Acta Hydrobiol. 23(2):183-190.

Lamb, R.J. and W.J. Turnock. 1982. Economics of insecticidal control of flea beetles (Coleoptera:Chrysomelidae) attacking rape in Canada. Can. Entomol. 114:827-840.

Lambert, S.M.. 1967. Functional relationship between sorption in soil and chemical structure. J. Agric. Food Chem. 15(4):572-576.

Lands Directorate. 1973. The Canada land inventory. Land capability classification for wildlife. Report no. 7. Environment Canada. 29 p.

Lands Directorate. 1976. Agriculture, 1965 land use cross tabulation. P.E.I./N.S./N.B./Que./Ont./Man./Sask./Alta.. Publication no. R002020. Environment Canada.

Lands Directorate. 1978a. The Canada land inventory. Objectives, scope and organization. Report no. 1 (first revision). Environment Canada. 61 p.

Lands Directorate. 1978b. Primary agriculture classes by waterfowl classification. Man./Sask./Alta.. Publication no. R002090. Environment Canada.

Lands Directorate. 1980. Waterfowl capability by province, census division breakdown. Land and Water. Man./Sask./Alta./B.C.. Publication no. R002250. Environment Canada.

Lathwell, D.J., D.R. Bouldin and E.A. Goyette. 1973. Growth and chemical composition of aquatic plants in twenty artificial wildlife marshes. New York Fish and Game Journal 20:108-128.

Lawrence, J.M.. 1962. Aquatic herbicide data. Agriculture Handbook no. 231. U.S. Dept. of Agriculture. 133 p.

Lee, G.F., E. Bentley and R. Amundson. 1975. Effects of marshes on water quality. In Coupling of Land and Water Systems, A.D. Haster (Ed.). Ecological studies vol. 10. Springer-Verlag. pp. 105-127.

Lehman, J.T.. 1980. Nutrient recycling as an interface between algae and grazers in freshwater communities. In Evolution and Ecology of Zooplankton Communities, W.C. Kerfoot (Ed.). American Society of Limnology and Oceanography Special Symposium vol. 3. pp. 251-263.

Leitch, W.G. and R.M. Kaminski. 1985. Long-term wetland-waterfowl trends in Saskatchewan grassland. J. Wildl. Manage. 49(1):212-222.

Lichtenstein, E.P., T.T. Liang and B.N. Anderegg. 1973. Synergism of insecticides by herbicides. Science 181:847-849.

Lieffers, V.J. and J.M. Shay. 1981. The effects of water level on the growth and reproduction of Scirpus maritimus var. paludosus. Can. J. Bot. 59:118-121.

Lightbody, J.P. and C.D. Ankney. 1984. Seasonal influences on the strategies of growth and development of canvasback and lesser scaup ducklings. Auk 101:121-133.

Linden, E., B.E. Bengtsson, O. Svanberg and G. Sundstrom. 1979. The acute toxicity of 78 chemicals and pesticide formulations against two brackish water organisms, the bleak (Alburnus alburnus) and the harpacticoid Nitocra spinipes. Chemosphere 11/12:843-851.

Liss, P.S. and P.G. Slater. 1974. Flux of gasses across the air-sea interface. Nature 247:181-184.

Livezey, B.C.. 1980. Effects of selected observer-related factors on fates of duck nests. Wildl. Soc. Bull. 8(2):123-127.

Lockhart, W.L., D.A. Metner, B.G. Billeck, G.P. Rawn and D.C.G. Muir. 1984. Bioaccumulation of some forestry pesticides by fish and aquatic plants. In Proc. Symp. Implications of Chemical and Biological Agents in Forestry Application, W.Y. Garner and J. Harvey Jr. (Eds.). Amer. Chem. Soc. no. 238. pp. 297-318.

Lokemoen, J.T., H.F. Duebbert and D.E. Sharp. 1984. Nest spacing, habitat selection and behavior of waterfowl on Miller Lake Island, North Dakota. J. Wildl. Manage. 48(2):309-321.

Loos, M.A.. 1975. Phenoxyalkanoic acids. In Degradation of Herbicides, P.C. Kearney and D.D. Kaufman (Eds.). Marcel Dekker, New York. pp. 1-49.

Loubaresse, J.P., V. Labonne, H. Jolie and A. Van Offeren. 1977. Mededelingen van de Faculteit Lanbouwetenschappen. Rijksuniversiteit Gent 42:1825-1838.

Low, J.B.. 1941. Nesting of the ruddy duck in Iowa. Auk 58:506-517.

Ludwig, P.D. and J.C. McNeill. 1966. Results of laboratory and field tests with Dursban insecticide for mosquito control. Mosq. News 26(3):344-351.

Ludwig, P.D., H.J. Dishburger, J.C. McNeill, W.D. Miller and J.R. Rice. 1968. Biological effects and persistence of Dursban insecticide in a salt marsh habitat. J. Econ. Entomol. 61(3):626-633.

Lyman, W.L., W.F. Reehl and D.H. Rosenblatt (Eds.). 1982. Handbook of chemical property estimation methods. Environmental behavior of organic compounds. McGraw-Hill Book Company, New York. 977 p.

Lynch, J.J.. 1984. Escape from mediocrity. Wildlfowl 35:5-13.

Lynch-Stewart, P.. 1984. Land use change on wetlands in southern Canada: review and bibliography. Lands Directorate Working Paper no. 26. Environment Canada. 115 p.

Lynn, G.E.. 1965. A review of toxicological information on Tordon herbicides. Down to Earth 20(4):6-8.

Macalady, D.L. and N.L. Wolfe. 1983. New perspectives on the hydrolytic degradation of the organophosphorothioate insecticide chlorpyrifos. J. Agric. Food Chem. 31:1139-1147.

Macan, T.T. and A. Kitching. 1972. Some experiments with artificial substrata. Verh. Internat. Verein. Limnol. 18:213-220.

Macek, K.J., D.F. Walsh, J.W. Hogan and D.D. Holz. 1972. Toxicity of insecticide Dusban to fish and aquatic invertebrates in ponds. Trans. Am. Fish. Soc. 101(3):420-427.

Macek, K.J., K.S. Buxton, S. Sauter, S. Gnilka and J.W. Dean. 1976. Chronic toxicity of atrazine to selected aquatic invertebrates and fishes. Environmental Research Laboratory, Duluth. U.S. Environmental Protection Agency, Duluth, Minnesota. EPA 600/3-76-047.

Mackay, D.. 1982. Correlation of bioconcentration factors. Environ. Sci. Technol. 16(5):274-278.

Mackay, D. and S. Paterson. 1982. Fugacity models for predicting environmental behavior of chemicals. Report to Environment Canada under contract OSU 81-00163.

Maier-Bode, H. and K. Hartel. 1981. Linuron and monolinuron. Residue Rev. 77:1-352.

Manitoba Agriculture. 1983. Guide to chemical weed control. Agdex 641. 115 p.

Manitoba Agriculture. 1984. 1984 Manitoba insect control guide. 47 p.

Manitoba Agriculture. 1985. Guide to chemical weed control. Agdex 641.

Marcström, V.. 1966. Mallard ducklings (*Anas platyrhynchos* L.) during the first days after hatching. A physiological study with ecological considerations and a comparison with capercaillie chicks (*Tetrao urogallus* L.). Vittrevy 4(5):342-370.

Marganian, V.M. and W.J. Wall Jr.. 1972. Dursban and diazinon residues in biota following treatment of intertidal plots on Cape Cod, 1967-69. Pestic. Monit. J. 6(3):160-165.

Marriage, P.B., A.A. Hamill and F.G. von Stryk. 1978. Response of soybean plants to metribuzin and interaction with atrazine residues. J. Environ. Sci. Health B13:287-297.

Marshall, E.J.P.. 1984. The ecology of a land drainage channel. II. Biology, chemistry and submerged weed control. Water Res. 18(7):817-825.

Martin, A.C., N. Hotchkiss, F.M. Uhler and W.S. Bourn. 1953. Classification of wetlands of the United States. U.S. Fish and Wildlife Service Special Science Report Wildlife no. 20. 14 p.

Martin, A.C. and F.M. Uhler. 1939. Food of game ducks in the United States and Canada. U.S. Dept. of Agriculture Technical Bulletin no. 634. 156 p.

Martin, F.W., R.S. Pospahala and J.D. Nichols. 1979. Assessment and population management of North American migratory birds. *In* Environmental Biomonitoring, Assessment, Prediction and Management. Certain Case Studies and Related Quantitative Issues, J. Cairns Jr., P.P. Ganapati and W.E. Waters (Eds.). International Cooperative Publishing House, Maryland. pp. 187-239.

Martin, H. and C.R. Worthing. 1977. Pesticide manual. Fifth edition. British Crop Protection Council. Boots Company Ltd., Nottingham, England.

Martz, G.F.. 1967. Effects of nesting cover removal on breeding puddle ducks. J. Wildl. Manage. 31(2):236-247.

Mason, H.L.. 1957. A flora of California marshes. University of California Press. 878 p.

Mauck, W.L., F.L. Mayer Jr. and D.D. Holz. 1976. Simazine residue dynamics in small ponds. Bull. Environ. Contam. Toxicol. 16:1-8.

May, B.E., R.S. Hestland and J.M. Van Dyke. 1973. Comparative effects of diquat plus copper sulfate on aquatic organisms. Weed Sci. 21(3):249-253.

Maybank, J. and K. Yoshida. 1969. Delineation of herbicide drift hazards on the Canadian prairies. Trans. ASAE 12:759-762.

Maybank, J., K. Yoshida, S.R. Shewchuk and R. Grover. 1976. Comparison of swath deposit and drift characteristics of ground-rig and aircraft spray systems; report of the 1975 field trials. Report of the Saskatchewan Research Council, January 1976.

Maybank, J., K. Yoshida, S.R. Shewchuk and R. Grover. 1977. Spray drift and swath deposit pattern from agricultural pesticide application; report of the 1976 field trial program. Report of the Saskatchewan Research Council, January 1977, P77-1.

Maybank, J., K. Yoshida, S.R. Shewchuk and R. Grover. 1978a. Spray drift behavior of aerially-applied herbicide; report of the 1977 field trials. Report of the Saskatchewan Research Council, March 1978, P78-2.

Maybank, J., K. Yoshida and R. Grover. 1978b. Spray drift from agricultural pesticide applications. J. Air Pollut. Control Assoc. 28(10):1009-1014.

Mayes, M.A. and D.C. Dill. 1984. The acute toxicity of picloram, picloram potassium salt, and picloram triisopropanolamine salt to aquatic organisms. Environ. Toxicol. Chem. 3:263-269.

Mayeux, H.S. Jr., C.W. Richardson, R.W. Bovey, E. Burnett, M.G. Merkle and R.E. Meyer. 1984. Dissipation of picloram in storm runoff. J. Environ. Qual. 13(1):44-49.

Mayfield, H.. 1961. Nesting success calculated from exposure. Wilson Bull. 73(3):255-261.

McClellan, J.B., L. Jersak and C.L.A. Hutton. 1968. A guide to the classification of land use for the Canada land inventory. Unpublished Report 302-11-1-2. Dept. of Energy, Mines and Resources, Ottawa, Canada. 18 p.

McCuaig, J.D. and E.W. Manning. 1982. Agricultural land-use change in Canada: process and consequences. Land use in Canada no. 21. Lands Directorate. Environment Canada. 213 p.

McDonald, M.E.. 1955. Causes and effects of a die-off of emergent vegetation. J. Wildl. Manage. 19:24-35.

McLeese, D.W., C.D. Metcalfe and V. Zitko. 1980. Lethality of permethrin, cypermethrin and fenvalerate to salmon, lobster and shrimp. Bull. Environ. Contam. Toxicol. 25:950-955.

Means, J.C. and R. Wijayaratne. 1982. Role of natural colloids in the transport of hydrophobic pollutants. Science 215:968-970.

Meikle, R.W.. 1972. Hydrolysis rate of chlorpyrifos in buffered distilled water. Agricultural Research. Dow Chemical Company. Walnut Creek, California. Unpublished Report.

Melnikov, N.N.. 1971. Chemistry of pesticides. Residue Rev. 36:1-480.

Merna, J.W. and P.J. Eisele. 1973. Effects of methoxychlor on aquatic birds. U.S. Environmental Protection Agency, Ecological Research Series. EPA-R3-73-046.

Merriam, G.. 1975. Aspen parkland slough habitats in Alberta. Unpublished Report. Canadian Wildlife Service. Western Region. 68 p.

Merriam, G.. 1978. Changes in aspen parkland habitat bordering Alberta sloughs. Can Field-Nat. 92(2):109-122.

Messersmith, C.G., O.C. Burnside and T.L. Lavy. 1971. Biological and non-biological dissipation of trifluralin from soil. Weed Sci. 19:288-290.

Metcalf, F.P.. 1931. Wild-duck foods of North Dakota. U.S.D.A. Tech. Bull. 221. 72 p.

Metcalf, R.L.. 1980. Changing role of insecticides in crop protection. Ann. Rev. Entomol. 25:219-256.

Metcalf, R.L. and J.R. Sanborn. 1975. Pesticides and environmental quality in Illinois. Ill. Nat. Hist. Surv. Bull. 31(9):381-436.

Metras, L.. 1986. Migratory birds killed in Canada during the 1985 season. Canadian Wildlife Service Progress Note No. 166. 42 p.

Meyboom, P.. 1966. Unsteady groundwater flow near a willow ring in hummocky moraine. J. Hydrol. 4:38-62.

Meyer, F.P.. 1965. The experimental use of Guthion^R as a selective fish eradicator. Trans. Am. Fish. Soc. 94:203-209.

Miles, J.R.W. and C.R. Harris. 1978. Insecticide residues in water, sediment, and fish of the drainage system of the Holland Marsh, Ontario, Canada, 1972-75. J. Econ. Entomol. 71(1):125-131.

Millar, J.B.. 1969. Some characteristics of wetland basins in central and southwestern Saskatchewan. In Saskatoon Wetland Seminar. Canadian Wildlife Service Report Series no. 6. pp. 73-101.

Millar, J.B.. 1971. Shoreline-area ratio as a factor in rate of water loss from small sloughs. J. Hydrol. 14:259-284.

Millar, J.B.. 1973. Vegetation changes in shallow marsh wetlands under improving moisture regime. Can. J. Bot. 51:1443-1457.

Millar, J.B.. 1976. Wetland classification in western Canada. Canadian Wildlife Service Report Series no. 37. 38 p.

Millar, J.B.. 1982. Classification of wetlands on air/ground comparison transects in the prairie provinces. Part I. Waterfowl strata 30 to 33. Saskatchewan. Canadian Wildlife Service Project Report. Saskatoon. 74 pp.

Millar, J.B.. 1983. Classification of wetlands on air/ground comparison transects in the prairie provinces. Part II. Waterfowl strata 34 to 40. Southeastern Saskatchewan and Manitoba. Canadian Wildlife Service Project Report. Saskatoon. 82 p.

Millar, J.B.. 1984. Classification of wetlands on air/ground comparison transects in the prairie provinces. Part III. Waterfowl strata 26 to 29 Alberta. Canadian Wildlife Service Project Report. Saskatoon. 77 p.

Miller, H.W. and D.H. Johnson. 1978. Interpreting the results of nesting studies. J. Wildl. Manage. 42(3):471-476.

Milonski, M.. 1958. The significance of farmland for waterfowl nesting and techniques for reducing losses due to agricultural practices. Trans. N. Am. Wildl. Conf. 23:215-228.

Mittelbach, G.G.. 1981. Patterns of invertebrate size and abundance in aquatic habitats. Can. J. Fish. Aquat. Sci. 78:896-904.

Miyamoto, J.. 1976. Degradation, metabolism and toxicity of synthetic pyrethroids. Environ. Health Perspect. 14:15-28.

Miyamoto, J. and N. Mikami. 1983. Degradation of pyrethroid insecticides in the field. In Pesticide Chemistry: Human Welfare and the Environment. Volume 2. Natural Products, J. Miyamoto and P.C. Kearney (Eds.). Pergamon Press. pp. 193-200.

Mohsen, Z.H. and M.S. Mulla. 1981. Toxicity of blackfly larvicidal formulations to some aquatic insects in the laboratory. Bull. Environ. Contam. Toxicol. 26:696-703.

Morris, K. and R. Jarman. 1981. Evaluation of water quality during herbicide applications to Kerr Lake, OK. J. Aquat. Plant Manage. 19:15-18.

Morris, R.F. and B.G. Penney. 1971. Persistence of linuron residues in soils at Kelligrews, Newfoundland. Can. J. Plant Sci. 51:242-245.

Morrison, M.L. and E.C. Meslow. 1982. Impacts of forest herbicides on wildlife: toxicity and habitat alteration. 48th Trans. N. Am. Wildl. Conf. pp. 175-185.

Morrison, M.L. and E.C. Meslow. 1984. Effects of the herbicide glyphosate on bird community structure, Western Oregon. For. Sci. 30(1):95-106.

Moulding, H.C.. 1983. Impact of drought on wildlife. Unpublished Report. Drought Workshop, Regina 29-30 August 1983. Ducks Unlimited Canada. 13 p.

Moyle, J.B.. 1961. Aquatic invertebrates as related to larger water plants and waterfowl. Minnesota Dept. of Conservation. Division of Game and Fish. Investigational Report no. 233. 24 p.

Muir, D.C.G.. 1986. Persistence and transformations in water and sediment. In Environmental Chemistry of Herbicides. Volume 2, R. Grover (Ed.). CRC Press. (in press)

Muir, D.C.G., G.P. Rawn, B.E. Townsend, W.L. Lockhart and R. Greenhalgh. 1985. Bioconcentration of cypermethrin, deltamethrin, fenvalerate and permethrin by Chironomus tentans larvae in sediment and water. Environ. Toxicol. Chem. 4:51-61.

Muir, D.C.G., B.E. Townsend and W.L. Lockhart. 1983. Bioavailability of six organic chemicals to Chironomus tentans larvae in sediment and water. Environ. Toxicol. Chem. 2:269-281.

Muirhead-Thomson, R.C.. 1970. The potentiating effect of pyrethrins and pyrethroids on the action of organophosphorus larvicides in Simulium control. Trans. Soc. Trop. Med. Hyg. 64(6):895-906.

Muirhead-Thomson, R.C.. 1971. Pesticides and freshwater fauna. Academic Press. London, New York. 231 p.

Muirhead-Thomson, R.C.. 1978a. Lethal and behavioral impact of permethrin (NRDC 143) on selected stream macroinvertebrates. Mosq. News 38(2):185-190.

Muirhead-Thomson, R.C.. 1978b. Relative susceptibility of stream macroinvertebrates to temephos and chlorpyrifos, determined in laboratory continuous-flow systems. Arch. Environ. Contam. Toxicol. 7:129-137.

Mulla, M.S., H.A. Darwazeh and M.S. Dhillon. 1980. New pyrethroids as mosquito larvicides and their effects on non-target organisms. Mosq. News 40(1):6-12.

Mulla, M.S., H.A. Navvab-Gojrati and H.A. Darwazeh. 1978. Biological activity and longevity of new synthetic pyrethroids against mosquitoes and some non-target insects. Mosq. News 38(1):90-96.

Mulla, M.S., R.L. Norland, W.E. Westlake, B. Dell and J. St-Amant. 1973. Aquatic midge larvicides, their efficacy and residues in water, soil, and fish in a warm-water lake. Environ. Entomol. 2(1):58-65.

Naqvi, S.M., V.O. Davis and R.M. Hawkins. 1985. Percent mortalities and LC_{50} values for selected microcrustaceans exposed to Triflen^R, Cutrine-plus^R and MSMA herbicides. Bull. Environ. Contam. Toxicol. 35:127-132.

National Farmers Union (NFU). 1983. National Farmer's Union submission to the cabinet committee on water concerns. Unpublished Document. Regina. 30 p.

Nesbitt, H.J. and J.R. Watson. 1980a. Degradation of the herbicide 2,4-D in river water. I. Description of study area and survey of rate determination factors. Water Res. 14:1683-1688.

Nesbitt, H.J. and J.R. Watson. 1980b. Degradation of the herbicide 2,4-D in river water. II. The role of suspended sediment, nutrients and water temperature. Water Res. 14:1689-1694.

Newbold, C.. 1974. The ecological effects of the herbicide dichlobenil within pond ecosystems. Proc. Eur. Weed Res. Counc. Symp. Aquat. Weeds, 4th, Wein. pp. 37-52.

Newbold C.. 1975. Herbicides in aquatic systems. Biol. Conserv. 7:97-118.

Newbold, C., M.C. French and C. Brown. 1974. Ecological effects of the herbicides diquat and dichlobenil within pond ecosystems. Rep. Monks Wood Exp. St. 1972-1973. pp. 29-31.

Newman, J.F. and J.M. Way. 1966. Some ecological observations in the use of paraquat and diquat as aquatic herbicide. Proc. 8th Brit. Weed Cont. Conf.. pp. 582-585.

Nicholaichuk, W. and R. Grover. 1983. Loss of fall-applied 2,4-D in spring runoff from a small agricultural watershed. J. Environ. Qual. 12(3):412-414.

Niering, W.A. and R.H. Goodwin. 1974. Creation of relatively stable shrublands with herbicides: arresting "succession" on rights-of-way and pastureland. Ecology 55:784-795.

Norris, L.A.. 1967. Chemical brush control and herbicide residues in the forest environment. In Proc. Symp. Herbicides and Vegetation Management in Forests, Ranges and Non-crop Lands. Oregon State University. pp. 103-123.

Norris, L.A.. 1981. The movement, persistence, and fate of the phenoxy herbicides and TCDD in the forest. Residue Rev. 80:65-135.

Noyes, J.H. and R.L. Jarvis. 1985. Diet and nutrition of breeding female redhead and canvasback ducks in Nevada. J. Wildl. Manage. 49(1):203-211.

NRCC. 1974. Picloram: the effects of its use as a herbicide on environmental quality. NRC Associate Committee on Scientific Criteria for Environmental Quality. NRCC 13684. National Research Council of Canada. 128 p.

NRCC. 1978a. Ecotoxicology of chlorpyrifos. NRC Associate Committee on Scientific Criteria for Environmental Quality. NRCC 16079. National Research Council of Canada. 314 p.

NRCC. 1978b. Phenoxy herbicides, their effects on environmental quality. NRC Associate Committee on Scientific Criteria for Environmental Quality. NRCC 16075. National Research Council of Canada. 440 p.

NRCC. 1979. Carbofuran: criteria for interpreting the effects of its use on environmental quality. NRC Associate Committee on Scientific Criteria for Environmental Quality. NRCC 16740. National Research Council of Canada. 191 p.

NRCC. 1981. A screen for the relative persistence of lipophilic organic chemicals in aquatic ecosystems. An analysis of the role of a simple computer model in screening. NRC Associate Committee on Scientific Criteria for Environmental Quality. NRCC 18570. National Research Council of Canada. 300 p.

NRCC. 1983. Impact assessments in lotic environments. Methoxychlor. NRC Associate Committee on Scientific Criteria for Environmental Quality. NRCC 20645. National Research Council of Canada. 130 p.

NRCC. 1986. Pyrethroids: their effects on aquatic and terrestrial ecosystems. NRC Associate Committee on Scientific Criteria for Environmental Quality. National Research Council of Canada. 295 p.. (in press)

Nudds, T.D. and J.N. Bowlby. 1984. Predator-prey size relationships in North American dabbling ducks. Can. J. Zool. 62:2002-2008.

O'Brien, R.D. and C.E. Dannelley. 1965. Penetration of insecticide through rat skin. J. Agric. Food Chem. 13:245-247.

Odeyemi, O.. 1982. Relative stability of carbaryl in tropical ecosystems. Environ. Pollut. A29:207-213.

OECD. 1981. OECD guidelines for testing of chemicals. The Organization for Economic Cooperation and Development.

Oetting, R.B. and C.C. Dixon. 1975. Waterfowl nest densities and success at Oak Hammock Marsh, Manitoba. Wildl. Soc. Bull. 3(4):166-171.

Ohkawa, H., K. Nambu, H. Inui and J. Miyamoto. 1978. Metabolic fate of fenvalerate (sumicidin) in soil and microorganism. J. Pestic. Sci. 3:129-141.

Ohno, N., K. Fujimoto, Y. Okuno, T. Mizutani, M. Hirano, N. Itaya, T. Honda and H. Yoshioka. 1976. 2-arylalkanoates, a new group of synthetic pyrethroid esters not containing cyclopropanecarboxylates. Pestic. Sci. 7:241-247.

Okland, J.. 1964. The eutrophic Lake Borrevann (Norway), an ecological study on shore and bottom fauna with special reference to gastropods, including a hydrographic survey. Folia Limnol. Scand. 13. 337 p.

Olsen, A.R. and S.E. Wise. 1982. Frequency analysis of pesticide concentrations for risk assessment (FRANCO MODEL). U.S. Environmental Protection Agency. Athens, Georgia.EPA 600/3-82-044.

Olson, D.P.. 1964. A study of canvasback and redhead breeding populations, nesting habitats and productivity. Ph.D. Thesis, University of MInnesota, Minneapolis, Minnesota. 100 p.

O'Neal, S.W. and C.A. Lembi. 1983. Effect of simazine on photosynthesis and growth of filamentous algae. Weed Sci. 31:899-903.

O'Neill, R.V., S.M. Bartell and R.H. Gardner. 1983. Patterns of toxicological effects in ecosystems: a modeling study. Environ. Toxicol. Chem. 2:451-461.

O'Neill, R.V., R.H. Gardner, L.W. Barnthouse, G.W. Suter, S.G. Hildebrand and C.W. Gehrs. 1982. Ecosystem risk analysis: a new methodology. Environ. Toxicol. Chem. 1:167-177.

Onishi, Y., S.M. Brown, A.R. Olsen, M.A. Parkhurst, S.E. Wise and W.H. Walters. 1982. Methodology for overland and instream migration and risk assessment of pesticides. U.S. Environmental Protection Agency, Athens, Georgia. EPA 600/3-82-024.

Onishi, Y. and S.E. Wise. 1982. Users manual for the instream sediment contaminant transport model, SERATRA. U.S. Environmental Protection Agency, Athens, Georgia. EPA 600/3-82-055.

Parkhurst, M.A., Y. Onishi and A.R. Olsen. 1981. A risk assessment of toxicants to aquatic life using environmental exposure estimates and laboratory toxicity data. In Aquatic Toxicology and Hazard Assessment, D.R. Branson and K.L. Dickson (Eds.). American Society for Testing and Materials. ASTM STP 737. pp. 59-71.

Parr, J.F. and S. Smith. 1973. Degradation of trifluralin under laboratory conditions and soil anaerobiosis. Soil Sci. 115(1):55-63.

Paterson, C.G. and C.H. Fernando. 1969. The effect of winter drainage on reservoir benthic fauna. Can. J. Zool. 47:589-595.

Patterson, J.H.. 1973. Ecological considerations of herbicide use in Alberta. Canadian Wildlife Service Manuscript Report no. 26.

Patterson, J.H.. 1978. Canadian waterfowl management plan. Unpublished Report. Canadian Wildlife Service. Environment Canada. 88 p.

Peck, D.E., D.L. Corwin and W.J. Farmer. 1980. Adsorption-desorption of diuron by freshwater sediments. J. Environ. Qual. 9:101-106.

Peden, D., S. Baker and G. Shaw. 1982. Agricultural chemical data review for the prairies and some implications for environmental research. Unpublished Report. Environment Canada. Edmonton. 42 p.

Pehrsson, O.. 1979. Feeding behavior, feeding habitat utilization, and feeding efficiency of mallard ducklings (Anas platyrhynchos L.) as guided by a domestic duck. Viltrevy (Swedish Wildlife) 10(8):193-215.

Pehrsson, O.. 1984. Relationships of food to spatial and temporal breeding strategies of mallard in Sweden. J. Wildl. Manage. 48(2):322-339.

Pennak, R.W.. 1978. Freshwater invertebrates of the United-States. John Wiley and sons. 803 p.

Penney, J.G. and E.D. Bailey. 1970. Comparison of energy requirements of fledgling black ducks and coots. J. Wildl. Manage. 34(1):105-114.

Perret, N.G.. 1962. The spring and summer foods of the common mallard (Anas platyrhynchos L.) in southcentral Manitoba. M.Sc. Thesis, University of British Columbia, Vancouver, British Columbia. 82 p.

Peterson, G.E.. 1984. The farmer as a customer of custom aerial spray services. In Proc. Symp. on the Future Role of Aviation in Agriculture, P.W. Voisey (Ed.). Associate Committee on Agricultural and Forestry Aviation. AFA-TN-17. NRC no. 23504. pp. 243-247.

Peverly, J.H. and R.L. Johnson. 1979. Nutrient chemistry in herbicide-treated ponds of differing fertility. J. Environ. Qual. 8(3):294-300.

Pip, E. and J.M. Stewart. 1976. The dynamics of two aquatic plant-snail associations. Can. J. Zool. 54:1192-1205.

Pokorny, J., J. Mentberger, B. Losos, P. Hartman and J. Hetesa. 1971. Changes in hydrochemical and hydrobiological relations occurring when Elodea canadensis was controlled with paraquat. In Proc. Eur. Weed Res. Coun. 3rd Int. Symp. Aquatic Weeds. pp. 217-229.

Polunin, N.V.C.. 1984. The decomposition of emergent macrophytes in fresh water. Adv. Ecol. Res. 10:115-166.

Pospahala, R.S, D.R. Anderson and C.J. Henny. 1974. Population ecology of the mallard. II. Breeding habitat conditions, size of the breeding populations, and production indices. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife. Resource Publication no. 115. 73 p.

Poston, H.J.. 1974. Home range and breeding biology of the shoveler. Canadian Wildlife Service Report Series no. 25. 49 p.

Potts, G.R.. 1974. The grey partridge: problems of quantifying the ecological effects of pesticides. Proc. Int. Congr. Game Biol. 11:405-413.

Potts, G.R.. 1977. Population dynamics of the grey partridge: overall effects of herbicides and insecticides on chick survival rates. Proc. Int. Congr. Game Biol. 13:203-211.

Potts, G.R.. 1985. Monitoring changes in the cereal ecosystem. Unpublished Manuscript.

Presing, M.. 1981. On the effects of Dikonirt (sodium salt of 2,4-dichlorophenoxi-acetic acid) on the mortality and reproduction of Daphnia magna. Hydrobiologia 83:511-516.

Pritchard, M.K. and E.H. Stobbe. 1980. Persistence and phytotoxicity of dinitroaniline herbicides in Manitoba soils. Can J. Plant Sci. 60:5-11.

Pruss, S.W. and E.R. Higgins. 1975. Effects of low levels of simazine on plankton algae in a non-stratified lake. Proc. Northeast. Weed Sci. Soc. 29:63-67.

Quinn, S.A., N.F. Gardarelli and E.O. Gangstad. 1977. Aquatic herbicide chronicity. J. Aquat. Plant Manage. 15:74-76.

Ragab, M.T.H.. 1974. Simazine persistence in soil and effect of its residue on crops. Can. J. Plant Sci. 54:713-716.

Ragab, M.T.H.. 1975. Residues of picloram in soil and their effects on crops. Can. J. Soil Sci. 55:55-59.

Ragab, M.T.H. and J.S. Leefe. 1972. Residues in soils and strawberries resulting from simazine applications. Can. J. Plant Sci. 52:147-149.

Rahman, A. and R. Ashford. 1973. Persistence of trifluralin under field conditions in Saskatchewan. Can. J. Plant Sci. 53:421-423.

Rakowski, P.W. and B.P. Chabot. 1983. Changes in land use in the Minnedosa district of southwestern Manitoba, an update on the Kiel-Hawkins transects. Unpublished Report. Canadian Wildlife Service, Winnipeg, Manitoba. Environment Canada. 15 p.

Rands, M.R.W.. 1985. Pesticide use on cereals and the survival of partridge chicks: a field experiment. J. Appl. Ecol. 22(1):49-54.

Rands, M.R.W. and N. Sotherton. 1985. Pesticides threaten British wildlife. New Scientist, 4th July: pp. 32.

Rao, P.S.C. and J.M. Davidson (Eds.). 1982. Retention and transformation of selected pesticides and phosphorus in soil-water systems: a critical review. Environmental Research Laboratories. U.S. Environmental Protection Agency, Athens, Georgia. PB 82-256884.

Rawash, I.A., I.A. Gaaboub, F.M. El-Gayar and A.Y. El-Shazli. 1975. Standard curves for nuvacron, malathion, sevin, DDT and kelthane tested against the mosquito Culex pipiens L. and the microcrustacean Daphnia magna Straus. Toxicology 4:133-144.

Rawn, G.P., D.C.G. Muir and G.R.B. Webster. 1983. Uptake and persistence of permethrin by fish, vegetation and hydrosoil. In Proc. 8th Annual Aquatic Toxicity Workshop, N.K. Kaushik and K.R. Solomon (Eds.). Can. Tech. Rep. Fish. Aquat. Sci. no. 1151. p. 195.

Rawn, G.P., G.R.B. Webster and G.M. Findlay. 1978. Effect of pool bottom substrate on residues and bioactivity of chlorpyrifos, against larvae of Culex torsalis (Diptera:Culicidae). Can. Entomol. 110:1269-1276.

Rawn, G.P., G.R.B. Webster and D.C.G. Muir. 1980. Bioactivity and degradation of permethrin in artificial pools. In Proc. 6th Annual Aquatic Toxicity Workshop, J.F. Klaverkamp, S.L. Leonard and K.E. Marshall (Eds.). Can. Tech. Rep. Fish. Aquat. Sci. no. 975. pp. 140-149.

Rawn, G.P., G.R.B. Webster and D.C.G. Muir. 1982. Fate of permethrin in model outdoor ponds (water, hydrosoil, photolysis, duckweed). J. Environ. Sci. Health B17:463-486.

Ree, H.I., J.C. Shim, H.K. Hong, J.S. Lee, H.W. Cho and C.L. Kim. 1981. Studies on control effects of pesticide applications against the vector mosquito larvae in rice fields in Korea. Korean J. Entomol. XI(2):39-45.

Reed, A.. 1975. Reproductive output of black ducks in the St-Lawrence estuary. J. Wildl. Manage. 39:243-255.

Reinecke, K.J.. 1979. Feeding ecology and development of juvenile black ducks in Maine. Auk 96:737-745.

Renne, D.S. and M.A. Wolf. 1979. Experimental studies of 2,4-D herbicide drift characteristics. Agric. Meteorol. 20:7-24.

Rennie, D.A., J.D. Beaton and R.A. Hedlin. 1980. The role of fertilizer nutrients in western Canadian development. Canada West Foundation Publication no. 80-139. Calgary. 47 p.

Rennie, D.A., E. de Jong, E.H. Halstead, R.G. Kachanoski, D.B. Wilkinson and R.P. Voroney. 1984. Innovative Acres 1984 Report. Saskatchewan Institute of Pedology Publication no. M75. 174 p.

Rettich, F.. 1977. The susceptibility of mosquito larvae to eighteen insecticides in Czechoslovakia. Mosq. News 37(2):252-257.

Rettich, F.. 1979. The toxicity of four synthetic pyrethroids to mosquito larvae and pupae (Diptera, Culicidae) in Czechoslovakia. Acta Entomol. Bohemoslov. 76:395-401.

Rettich, F.. 1980. The use of permethrin in pre-seasonal treatment of mosquito breeding sites. Folia Parasitologica (PRAHA) 27:265-268.

Richardson, C.J.. 1979. Primary productivity values in fresh water wetlands. *In* Wetland Functions and Values: the State of our Understanding, P.E. Greeson, J.R. Clark and J.E. Clark (Eds.). American Water Resources Association. pp. 131-145.

Ritcey, G., F.L. McEwen, H. Braun and R. Frank. 1981. Initial pesticide residues in relation to vapor pressure. Pestic. Sci. 12:614-618.

Roadhouse, F.E.B. and L.A. Birk. 1961. Penetration and persistence in soil of the herbicide 2-chloro-4,6-bis(ethylamino)-s-trizine(simazine). Can. J. Plant Sci. 41(2):252-260.

Roberts, J.R. and W.K. Marshall. 1980. Retentive capacity: an index of chemical persistence expressed in terms of chemical-specific and ecosystem-specific parameters. Envir. Toxicol. Saf. 4:158-171.

Robertson, E.B. and D.L. Bunting. 1976. The acute toxicity of four herbicides to 0-4 hour nauplii of *Cyclops vernalis* (Copepoda, Cyclopoida). Bull. Environ. Contam. Toxicol. 6(6):682-688.

Robinson, E. and L.L. Fox. 1978. 2,4-D herbicides in central Washington. J. Air Pollut. Control Assoc. 28(10):1015-1020.

Robson, T.O.. 1966. Some studies of the persistence of 2,4-D in natural surface waters in Britain. Proc. 8th Brit. Weed Control Conf.. pp. 594-597.

Rooke, J.B.. 1984. The invertebrate fauna of four macrophytes in a lotic system. Freshwater Biol. 14:507-513.

Rosene, W. Jr. and D.W. Lay. 1963. Disappearance and visibility of quail remains. J. Wildl. Manage. 27:139-142.

Rosine, W.N.. 1955. The distribution of invertebrates on submerged aquatic plant surfaces in Muskee Lake, Colorado. Ecology 36(2):308-314.

Roussel-Uclaf. 1982. Deltamethrin. Roussel Uclaf, Paris, France. 412 p.

Rubec, C.D.A. and P.C. Rump. 1984. Prime wetland land use monitoring activities and needs in southern Canada. Unpublished Report. Third Annual Western Provincial Conferences on Rationalization of Water and Soil Research and Management. Lands Directorate. Environment Canada.

Rubio-Moran, R., W.A. Rowley and J.R. Coats. 1981. Effect of permethrin and malathion on Aedes trivittatus and Culex pipiens pipiens. Iowa State J. Res. 55(3):235-244.

Sacher, R.M.. 1978. Safety of Roundup^R in the aquatic environment. Proc. Eur. Weed Res. Coun. 5th Symp. on Aquatic Weeds 5:315-322.

Sanders, H.O.. 1969. Toxicity of pesticides to the crustacean Gammarus lacustris. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife Technical Paper no. 25. 18 p.

Sanders, H.O.. 1970. Toxicities of some herbicides to six species of freshwater crustaceans. J. Water Pollut. Control Fed. 42(8) part 1:1544-1550.

Sanders, H.O.. 1972. Toxicity of some insecticides to four species of malacostracan crustaceans. U.S. Fish and Wildlife Service Technical Paper no. 66. 19 p.

Sanders, H.O. and O.B. Cope. 1966. Toxicities of several pesticides to two species of cladocerans. Trans. Am. Fish. Soc. 95:165-169.

Sanders, H.O. and O.B. Cope. 1968. The relative toxicities of several pesticides to naiads of three species of stoneflies. Limnol. Oceanogr. 13:112-117.

Sanders, H.O., M.T. Finley and J.B. Hunn. 1983. Acute toxicity of six forest insecticides to three aquatic invertebrates and four fishes. U.S. Fish and Wildlife Service Technical Paper no. 110. 5 p.

Sangha, G.K.. 1972. Environmental effects of carbamate insecticides as assayed in the model ecosystem, a comparison with DDT. Diss. Abstr. Int. 32:4650-B.

Santharam, K.R., B. Thayumanavan and S. Krishnaswamy. 1976. Toxicity of some insecticides to Daphnia carinata King, an important link in the food chain in the freshwater ecosystems. Indian J. Ecol. 3(1):70-73.

Saskatchewan Agriculture. 1983. Chemical weed control in cereal, oilseed, pulse and forage crops 1983. 40 p.

Saskatchewan Agriculture. 1984. Insect control on field crops 1984. 14 p.

Savidge, J.A.. 1978. Wildlife in a herbicide-treated Jeffrey pine plantation in eastern California. J. For. August 1978:476-478.

Scheffer, M., A.A. Achterberg and B. Beltman. 1984. Distribution of macroinvertebrates in a ditch in relation to the vegetation. Freshwater Biol. 14:367-370.

Schimmel, S.C., R.L. Garnas, J.M. Patrick Jr. and J.C. Moore. 1983. Acute toxicity, bioconcentration, and persistence of AC222,705, benthiocarb, chlorpyrifos, fenvalerate, methyl parathion and permethrin in the estuarine environment. J. Agric. Food Chem. 31(1):104-113.

Schladweiler, P. and J.P. Weigand. 1983. Relationships of endrin and other chlorinated hydrocarbon compounds on wildlife in Montana 1981-82. Report for the Wildlife Division, Montana Dept. of Fish, Wildlife and Parks. Helena, Montana.

Schober, U. and W. Lampert. 1977. Effects of sublethal concentrations of the herbicide atrazine on the growth and reproduction of Daphnia pulex. Bull. Environ. Contam. Toxicol. 17(3):269-277.

Schobert, E.E.. 1974. Ingested insecticide poisoning in waterfowl. J. Zoo. Anim. Med. 5:20.

Schultz, D.P. and E.W. Whitney. 1974. Monitoring 2,4-D residues at Loxahatchee National Wildlife Refuge. Pestic. Monit. J. 7(3/4):146-152.

Scifres, C.J., T.J. Allen, C.L. Leinweber and K.H. Pearson. 1973. Dissipation and phytotoxicity of dicamba residues in water. J. Environ. Qual. 2(2):306-309.

Scifres, C.J., H.G. McCall, R. Maxey and H. Tai. 1977. Residual properties of 2,4,5-T and picloram in sandy rangeland soils. J. Environ. Qual. 6:36-42.

Scorgie, H.R.A.. 1980. Ecological effects of the aquatic herbicide cyanatryn on a drainage channel. J. Appl. Ecol. 17:207-225.

Scott, B.F., D.S. Painter, E. Nagy, B.J. Dutka and W.D. Taylor. 1981. Fate and effects of 2,4-D formulations as herbicides in aquatic ecosystems. Part 1. National Water Research Institute. Environment Canada. 73 p.

Sculthorpe, C.D.. 1967. The biology of aquatic vascular plants. Edward Arnold Ltd., London. 610 p.

Serie, J.R. and G.A. Swanson. 1976. Feeding ecology of breeding gadwalls on saline wetlands. J. Wildl. Manage. 40(1):69-81.

Serns, S.L.. 1975. The effects of dipotassium endothall on the zooplankton and water quality of a small pond. Water Resour. Bull. 11(6):1221-1231.

Sethunathan, N.. 1972. Diazinon degradation in submerged soil and rice-paddy water. In Fate of Organic Pesticides in the Aquatic Environment, S.D. Faust (ed.). Advances in Chemistry Series 111. pp. 244-255.

Sharom, M.S., J.R.W. Miles, C.R. Harris and F.L. McEwen. 1980a. Behavior of 12 insecticides in soil and aqueous suspensions of soil and sediment. Water Res. 14:1095-1100.

Sharom, M.S., J.R.W. Miles, C.R. Harris and F.L. McEwen. 1980b. Persistence of 12 insecticides in water. Water Res. 14:1089-1093.

Shaw, S.P. and C.G. Fredine. 1956. Wetlands of the United-States. U.S. Fish and Wildlife Service Circular 39. 67 p.

Shaw, G.C., D.K. Smith, P.J. Sheehan and P. Mineau. 1984. Environmental concerns. In Proc. Symp. on the Future Role of Aviation in Agriculture, P.W. Voisey (Ed.). Associate Committee on Agricultural and Forestry Aviation. National Research Council of Canada. AFA-TN-17, NRC no. 23504. pp. 47-63.

Sheets, T.J., J.R. Bradley Jr. and M.D. Jackson. 1972. Contamination of surface and ground water with pesticides applied to cotton. Univ. North Carolina Water Resour. Res. Inst. Rep. no. 60. 63 p.

Shires, S.W.. 1983. The use of small enclosures to assess the toxic effects of cypermethrin on fish under field conditions. Pestic. Sci. 14:475-480.

Siegfried, W.R.. 1973. Summer food and feeding of the ruddy duck in Manitoba. Can. J. Zool. 51:1293-1297.

Siegfried, W.R.. 1976. Segregation in feeding behavior of four diving ducks in southern Manitoba. Can. J. Zool. 54:730-736.

Sigmon, C.F.. 1979. Influence of 2,4-D and 2,4,5-T on life history characteristics of Chironomus (Diptera:Chironomidae). Bull. Environ. Contam. Toxicol. 21:596-599.

Simpson, R.L. and D. Pimentel. 1972. Ecological effects of the aquatic herbicide Fenac on small ponds. Search Agric. 2(10):57 p.

Simpson-Lewis, W., J.E. Moore, N.J. Pocock, M.C. Taylor and H. Swan. 1979. Canada's special resource lands: a national perspective of selected land uses. Lands Directorate Map Folio no. 4. Environment Canada. 232 p.

Simsiman, G.V. and G. Chesters. 1976. Persistence of diquat in the aquatic environment. Water Res. 10:105-112.

Simsiman, G.V., T.C. Daniel and G. Chesters. 1976. Diquat and endothall: their fates in the environment. Residue Rev. 62:131-174.

Sirons, G.J., R. Frank and T. Sawyer. 1973. Residues of atrazine, cyanazine, and their phytotoxic metabolites in a clay loam soil. J. Agric. Food Chem. 21:1016-1020.

Sjoberg, K. and K. Danell. 1982. Feeding activity of ducks in relation to diel emergence of chironomids. Can. J. Zool. 60:1383-1387.

Skidmore, J.F.. 1974. Factors affecting the toxicity of pollutants to fish. Vet. Rec. 94:456-458.

Skipper, H.D., C.M. Gilmour and W.R. Furtick. 1967. Microbial versus chemical degradation of atrazine in soils. Proc. Soil Sci. Soc. Amer. 31:653-656.

Slagsvold, T.. 1977. Bird population changes after clearance of deciduous scrub. Biol. Conserv. 12:229-244.

Sloan, C.E.. 1970. Prairie potholes and the water table. U.S. Geological Survey Prof. Paper 700-B:227-231.

Sloan, C.E.. 1972. Ground-water hydrology of prairie potholes in North Dakota. U.S. Geological Survey Prof. Paper 585-C. 28 p.

Sloof, W.. 1983. Benthic macroinvertebrates and water quality assessment: some toxicological considerations. Aquat. Toxicol. 4:73-82.

Smies, M., R.H.J. Evers, F.H.M. Peijnenburg and J.H. Koeman. 1980. Environmental aspects of field trials with pyrethroids to eradicate tsetse fly in Nigeria. Ecotox. Environ. Saf. 4:114-128.

Smith, A.E.. 1971. Disappearance of triallate from field soils. Weed Sci. 19:536-537.

Smith, A.E.. 1972. Persistence of trifluralin in small field plots as analysed by a rapid gas chromatographic method. J. Agric. Food Chem. 20:829-831.

Smith, A.E.. 1975. Field persistence studies with herbicides in prairie soils. In Environmental Quality and Safety, vol. 3 (suppl.), O. Coulston and O. Korte (Eds.). Thieme, Stuttgart. pp. 266-270.

Smith, A.E.. 1979. Soil persistence experiments with (^{14}C)2,4-D in herbicidal mixtures, and field persistence studies with triallate and trifluralin both singly and combined. Weed Res. 19:165-170.

Smith, A.E.. 1982. Herbicides and the soil environment in Canada. Can. J. Soil Sci. 62:443-460.

Smith, A.E. and G.S. Emmond. 1975. Persistence of linuron in Saskatchewan soils. Can. J. Soil Sci. 55:145-148.

Smith, A.E. and B.J. Hayden. 1976. Field persistence studies with eight herbicides commonly used in Saskatchewan. Can. J. Plant Sci. 56:769-771.

Smith, A.E. and B.J. Hayden. 1982. Comparison of the persistence of EPTC, metribuzin and propanil in Saskatchewan field soils. Bull. Environ. Contam. Toxicol. 29(2):240-242.

Smith, A.E., R. Grover, G.S. Emmond and H.C. Korven. 1975. Persistence and movement of atrazine, bromacil, monuron, and simazine in intermittently-filled irrigation ditches. Can. J. Plant Sci. 55:809-816.

Smith, A.G.. 1971. Ecological factors affecting waterfowl production in the Alberta parklands. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife Resource Publication no. 98. 49 p.

Smith, A.G. and J.H. Stoudt. 1968. Ecological factors affecting waterfowl production in the Canadian parklands. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife. Unpublished Report. 323 p.

Smith, A.G., J.H. Stoudt and J.B. Gollop. 1964. Prairie potholes and marshes. In Waterfowl Tomorrow, J.P. Linduska and A.L. Nelson (Eds.). U.S. Fish and Wildlife Service. pp. 39-50.

Smith, C.N., R.A. Leonard, G.W. Langdale and G.W. Bailey. 1978. Transport of agricultural chemicals from small upland piedmont watersheds. U.S. Environmental Protection Agency. Environmental Protection Technology Series. EPA 600/3-78/056.

Smith, G.E. and B.G. Isom. 1967. Investigation of effects of large-scale applications of 2,4-D on aquatic fauna and water quality. Pestic. Monit. J. 1(3):16-21.

Smith, K., N.K. Kaushik, K.R. Solomon and G. Guest. 1980. Evaluation of impact of a synthetic pyrethroid, permethrin, on a lake ecosystem using limnocorrals. Can. Tech. Rep. Fish. Aquat. Sci. 975:10-11.

Smith, L.M. and J.A. Kadlec. 1985. Fire and herbivory in a Great Salt Lake marsh. Ecology 66(1):259-265.

Smith, S., T.E. Reagan, J.L. Flynn and G.H. Willis. 1983. Azinphosmethyl and fenvalerate runoff loss from a sugarcane-insect IPM system. J. Environ. Qual. 12(4):534-537.

Soderquist, C.J., J.B. Bowers and D.G. Crosby. 1977. Dissipation of molinate in a rice field. J. Agric. Food Chem. 25:940-945.

Soderquist, C.J. and D.G. Crosby. 1975. Dissipation of 4-chloro-2-methylphenoxyacetic acid (MCPA) in a rice field. Pestic. Sci. 6:17-33.

Solomon, K.R., J.Y. Yoo, D. Lean, N.K. Kaushik, K.E. Day and G.L. Stephenson. 1985. Evaluation of the impact of permethrin on zooplankton communities using limnocorrals. I. Permethrin dissipation. Can. J. Fish. Aquat. Sci. 42:70-76.

Southwood, T.R.E. and D.J. Cross. 1969. The ecology of the partridge. III. Breeding success and the abundance of insects in natural habitats. J. Anim. Ecol. 38(3):497-507.

Sowls, L.K.. 1955. Prairie ducks: a study of their behavior, ecology and management. The Stackpole Company, Harrisburg, Pennsylvania. 193 p.

Spehar, R.L., D.K. Tanner and B.R. Nordling. 1983. Toxicity of the synthetic pyrethroids, permethrin and AC222,705 and their accumulation in early life stages of fathead minnows and snails. Aquat. Toxicol. 3:171-182.

Spencer, E.Y.. 1982. Guide to the chemicals used in crop protection. Seventh edition. Research Branch. Agriculture Canada Publication no. 1093. 595 p.

Standing Senatorial Committee on Agriculture, Fisheries and Forestry. 1984. Soil at risk. Canada's eroding future. The Senate of Canada. 129 p.

Stankovic, A., M. Arsenovic, M. Dimitrijevic and B. Konstantinovic. 1982. A study of possibility of elimination of submersed vegetation in irrigation system by new herbicides. In 6th Internat. Symp. on Aquatic Weeds. European Weed Research Council. Novi Sad. pp. 185-192.

Stanley, R.A.. 1975. Interaction of calcium and 2,4-D on eurasian watermilfoil. Weed Sci. 23(3):182-184.

Statistics Canada. 1947-1977. Sales of pest control products by Canadian registrants. Catalogue 46-212 Annual.

Statistics Canada. 1951-1981. Census of Canada, Agriculture. Ottawa, Canada.

Stemeroff, M. and J.A. George. 1983. The benefits and costs of controlling destructive insects on onions, apples and potatoes in Canada. A report prepared by the Entomological Society of Canada. 96 p.

Stephenson, R.R.. 1982. Aquatic toxicology of cypermethrin. I. Acute toxicity to some freshwater fish and invertebrates in laboratory tests. Aquat. Toxicol. 2:175-185.

Stephenson, R.R. and D.F. Kane. 1984. Persistence and effects of chemicals in small enclosures in ponds. Arch. Environ. Contam. Toxicol. 13:313-326.

Stewart, D.K.R. and S.O. Gaul. 1977. Persistence of 2,4-D, 2,4,5-T and dicamba in a dykeland soil. Bull. Environ. Contam. Toxicol. 18:210-218.

Stewart, R.E. and H.A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife Resources Publication 92. 57 p.

Stewart, R.E. and H.A. Kantrud. 1973. Ecological distribution of breeding waterfowl populations in North Dakota. J. Wildl. Manage. 37(1):39-50.

Stewart, R.E. and H.A. Kantrud. 1974. Breeding waterfowl populations in the prairie pothole region of North Dakota. Condor 76(1):70-79.

Stone, W.B.. 1979. Poisoning of wild birds by organophosphate and carbamate insecticides. N.Y. Fish Game J. 26(1):37-47.

Stone, W.B.. 1980. Bird deaths caused by pesticides used on turf grass. In Proc. 34th Ann. New York State Turf Grass Conf. pp. 58-64.

Stone, W.B. and P.B. Gradoni. 1985. Wildlife mortality related to use of the pesticide diazinon. New York State Department of Environmental Conservation. 27 pp.

Stone, W.B. and H. Knoch. 1982. American brant killed on golf courses by diazinon. N.Y. Fish Game J. 29(1):95-96.

Stoudt, J.H.. 1971. Ecological factors affecting waterfowl production in the Saskatchewan parklands. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife Resource Publication no. 99. 58 p.

Stoudt, J.H.. 1982. Habitat use and productivity of canvasbacks in southwestern Manitoba, 1961-1972. U.S. Fish and Wildlife Service Special Scientific Report. Wildlife no. 248. 31 p.

Strang, C.A.. 1980. Incidence of avian predators near people searching for waterfowl nests. J. Wildl. Manage. 44(1):220-222.

Stratton, G.W.. 1984. Effects of the herbicide atrazine and its degradation products, alone and in combination, on phototrophic organisms. Arch. Environ. Contam. Toxicol. 13:35-42.

Stratton, G.W., R.E. Burrell, M.L. Kurp and C.T. Corke. 1980. Interactions between the solvent acetone and the pyrethroid insecticide permethrin on activities of the blue-green alga Anabaena. Bull. Environ. Contam. Toxicol. 24:562-569.

Stratton, G.W. and C.T. Corke. 1981. Interaction of permethrin with Daphnia magna in the presence and absence of particulate matter. Environ. Pollut. A24:135-144.

Stratton, G.W., A.L. Huber and C.T. Corke. 1980. The effect of pesticides and their metabolites, alone and in combination, on algal processes. In Proc. 6th Ann. Aquat. Tox. Workshop, J.F. Klaverkamp, S.L. Leonard and K.E. Marshall (Eds.). Can. Tech. Rep. Fish. Aquat. Sci. no 975. pp. 131-139.

Street, M.. 1977. The food of mallard ducklings in a wet gravel quarry, and its relation to duckling survival. Wildfowl 28:113-125.

Street, M.. 1978. The role of insects in the diet of mallard ducklings, an experimental approach. Wildfowl 29:93-100.

Sturkie, P.D.. 1954. Avian physiology. Ichaca, Comstock Pub. Ass.. 766 p.

Sugden, L.G.. 1964. Food and food energy requirements of wild ducklings. Canadian Wildlife Service Progress Report no 1880. 27 p.

Sugden, L.G.. 1971. Feeding activity of captive lesser scaup. Canadian Wildlife Service. Progress Note no. 24.

Sugden, L.G.. 1973. Feeding ecology of pintail, gadwall, American Wigeon and lesser scaup ducklings. Canadian Wildlife Service Report Series no. 24. 43 p.

Sugden, L.G.. 1978. Canvasback habitat use and production in Saskatchewan parklands. Canadian Wildlife Service Occasional Paper no. 34. 32 p.

Sugden, L.G.. 1982. Feeding ecology of pintail, gadwall, American wigeon and lesser scaup ducklings in southern Alberta. Canadian Wildlife Service Report Series no. 24. 48 p.

Sugden, L.G. and G.W. Beyersbergen. 1984. Farming intensity on waterfowl breeding grounds in Saskatchewan parklands. Wildl. Soc. Bull. 12(1):22-26.

Sugden, L.G. and G.W. Beyersbergen. 1985. Prediction of duck nest survival in conventional and zero-tilled fields. Canadian Wildlife Service Progress Notes no. 156. 6 p.

Sugden, L.G., E.A. Driver and M.C.S. Kingsley. 1981. Growth and energy consumption by captive mallards. Can. J. Zool. 59(8):1567-1570.

Sugden, L.G. and L.E. Harris. 1972. Energy requirements and growth of captive lesser scaup. Poultry Sci. 51:625-633.

Suter, G.W., II, D.S. Vaughan and R.H. Gardner. 1983. Risk assessment by analysis of extrapolation error: a demonstration for effects of pollutants on fish. Environ. Toxicol. Chem. 2:369-378.

Swann, R.L., D.A. Laskowski, P.J. McCall, K.V. Kuy and H.J. Dishburger. 1983. A rapid method for the estimation of the environmental parameters octanol/water partition coefficient, soil sorption constant, water to air ratio, and water solubility. Residue Rev. 85:17-28.

Swanson, G.A., G.L. Krapu and J.R. Serie. 1979. Foods of laying female dabbling ducks on the breeding grounds. In Waterfowl and Wetlands. An Integrated Review, T.A. Bookout (Ed.). Proc. Symp. 39th Midwest Fish and Wildlife Conf. pp. 47-57.

Swanson, G.A. and M.I. Meyer. 1977. Impact of fluctuating water levels on feeding ecology of breeding blue-winged teal. J. Wildl. Manage. 41:426-433.

Swanson, G.A., M.I. Meyer and V.A. Adomaitis. 1985. Foods consumed by breeding mallards on wetlands of south-central North Dakota. J. Wildl. Manage. 49(1):197-203.

Swanson, G.A., M.I. Meyer and J.R. Serie. 1974. Feeding ecology of breeding blue-winged teals. J. Wildl. Manage. 38(3):396-407.

Swanson, G.A. and A.B. Sargeant. 1972. Observation of nighttime feeding behavior of ducks. J. Wildl. Manage. 36(3):959-961.

Szeto, S.Y., H.R. MacCarthy, P.C. Oloffs and R.F. Sheperd. 1979. The fate of acephate and carbaryl in water. J. Environ. Sci. Health B14(6):635-654.

Tagatz, M.E. and J.M. Ivey. 1981. Effects of fenvalerate on field and laboratory-developed estuarine benthic communities. Bull. Environ. Contam. Toxicol. 27:256-267.

Tagatz, M.E., P.W. Borthwick, G.H. Cook and D.C. Coppage. 1974. Effects of ground application of malathion on salt-marsh environments in northwestern Florida. Mosq. News. 34(3):309-315.

Takken, W., F. Balk, R.C. Jansen and J.H. Koeman. 1978. The experimental application of insecticides from a helicopter for the control of riverine populations of Glossina tachinoides in West Africa. VI. Observations on side-effects. Pans 24:455-466.

Talent, L.G., G.L. Krapu and R.L. Jarvis. 1982. Habitat use of mallard broods in south central North Dakota. J. Wildl. Manage. 46:629-635.

Thilenius, J.F., D.R. Smith and G.R. Brown. 1974. Effect of 2,4-D on composition and production of an alpine plant community in Wyoming. J. Range Manage. 27(2):140-142.

Thompson, B.H. 1975. Testing the susceptibility of Simulium damnosum (Diptera:Simulidae) to insecticides in the laboratory and in the field. Bul. WHO 52(3):351-357.

Thomson, J.N.. 1974. Environmental contaminants inventory study no. 1. Statistics on the use of pest control products in Canada. Inland Waters Directorate. Report Series no. 35. Environment Canada. 54 p.

Thut, R.N.. 1969. Feeding habits of larvae of seven Rhyacophila species with notes on other life-history features. Ann. Am. Entomol. Soc. 62:894-98.

Todd, B.G.. 1984. Usage of wild oat herbicides across the prairies. In Wild Oat Symposium Proc. vol. 2 (Use and Mode of Action of Wild Oat Herbicides), A.E. Smith and A.I. Hsiao (Eds.). Canada Plains Research Centre, University of Regina. pp. 21-30.

Tomkins, D.J. and W.F. Grant. 1974. Differential response of 14 weed species to seven herbicides in two plant communities. Can. J. Bot. 52:525-533.

Tomkins, D.J. and W.F. Grant. 1977. Effects of herbicides on species diversity of two plant communities. Ecology 58:398-406.

Tooby, T.E., A.N. Thompson, R.J. Rycroft, I.A. Black and R.T. Hewson. 1981. A pond study to investigate the effects on fish and aquatic invertebrates of deltamethrin applied directly onto water. Aquatic Environment Protection 2. Fisheries Laboratory. Burnham-on-Crouch, Essex, England.

Torres, A.M.R. and L.M. O'Flaherty. 1976. Influence of pesticides on Chlorella, Chlorococcum, Stigeoclonium (Chlorophyceae), Tribonema, Vaucheria (Xanthophyceae) and Oscillatoria (Cyanophyceae). Phycologia 15(1):25-36.

Townsend, G.H.. 1966. A study of waterfowl nesting on the Saskatchewan River delta. Can. Field-Nat. 80:74-88.

Trauger, D.L. and J.H. Stoudt. 1978. Trends in waterfowl populations and habitats on study areas in Canadian parklands. Trans. N. Am. Wildl. Nat. Resour. Conf. 43:187-205.

Trichell, D.W., H.L. Morton and M.G. Merkle. 1968. Loss of herbicides in runoff water. Weed Sci. 16:447-449.

Triplett, G.B. Jr., B.J. Conner and W.M. Edwards. 1978. Transport of atrazine and simazine in runoff from conventional and no-tillage corn. J. Environ. Qual. 7(1):77-84.

Tucker, C.S. and C.E. Boyd. 1981. Relationships between pond sediments and simazine loss from waters of laboratory systems. J. Aquat. Plant Manage. 19:55-57.

Tucker, R.K. and D.G. Crabtree. 1970. Handbook of toxicity of pesticides to wildlife. U.S. Fish and Wildlife Service. Bureau of Sport Fisheries and Wildlife Resource Publication no. 84. 131 p.

Tucker, R.K. and J.S. Leitzke. 1979. Comparative toxicology of insecticides for vertebrate wildlife and fish. Pharmac. Theor. 6:167-220.

U.S.D.A.. 1979. The biologic and economic assessment of dimethoate. U.S. Dept. of Agriculture Technical Bulletin no. 1663. 279 p.

U.S.D.I.. 1968. Water quality criteria. Report of the National Tech. Admin. Comm. to the Secretary of the Interior, Federal Water Pollution Control Administration. U.S. Dept. of the Interior.

U.S.E.P.A.. 1973. Water quality criteria. U.S. Environmental Protection Agency. EPA R3-73-003.

U.S.E.P.A.. 1985. Diazinon support document. Office of Pesticide Programs. 66 p.

U.S.N.R.C.. 1982. Impacts of emerging agricultural trends on fish and wildlife habitat. National Academy Press. Washington, D.C.. 303 p.

Valentine, J.P. and S.W. Bingham. 1974. Influence of several algae on 2,4-D residues in water. Weed Sci. 22(4):358-363.

Valentine, J.P. and S.W. Bingham. 1976. Influence of algae on amitrole and atrazine residues in water. Can. J. Bot. 54:2100-2107.

Vallentyne, J.R.. 1952. Insect removal of nitrogen and phosphorus compounds from lakes. Ecology 33(4):573-577.

Vanden Born, W.H.. 1969. Picloram residues and crop production. Can. J. Plant Sci. 49:628-629.

Van der Valk, A.G. and C.B. Davis. 1978. Primary production of prairie glacial marshes. In Freshwater Wetlands: Ecological Processes and Management Potential, R.E. Good, D.F. Whigham and R.L. Simpson (Eds.). Academic Press. pp. 21-34.

Van der Weij, H.G.. 1966. Recent developments and investigations in the chemical control of aquatic weeds in the Netherlands. Proc. Brit. Weed Contr. Conf. 8:835-841.

Van Valin, C.C.. 1966. Persistence of 2,6-dichlorobenzonitrile in aquatic environments. Adv. Chem. Ser. 60:271-279.

Veith, G.D., D.L. DeFoe and B.V. Bergstedt. 1979. Measuring and estimating the bioconcentration factor of chemicals in fish. J. Fish. Res. Board 36:1040-1048.

Vermeer, K.. 1972. Variation in density of breeding ducks across the aspen parklands and grasslands of Canada. The Blue Jay, Sept. 1972: 154-158.

Verschueren, K.. 1983. Handbook of environmental data on organic chemicals. Second Edition. Van Nostrand Reinhold Company. New York, Cincinnati, Toronto, London, Melbourne. 1310 p.

Virtanen, M., M.L. Hattula and A.V. Arstila. 1979. Behavior and fate of 4-chloro-2-methylphenoxyacetic acid (MCPA) and 4,6-dichloro-o-creosol as studied in an aquatic-terrestrial model ecosystem. Chemosphere 7:431-442.

Voigts, D.K.. 1976. Aquatic invertebrate abundance in relation to changing marsh vegetation. Am. Midl. Nat. 95(2):313-327.

Voisey, P.W.. 1984. Foreward. In Proc. Symp. on the Future Role of Aviation in Agriculture, P.W. Voisey (Ed.). Associate Committee on Agricultural and Forestry Aviation. AFA-TN-17, NRC no 23504.

Wade, P.M.. 1981. The long term effects of aquatic herbicides on the macrophyte flora of freshwater habitats, a review. In Proc. Conf. Aquatic Weeds and their Control, 1981. Christ Church, Oxford, England. pp. 233-240.

Wade, P.M.. 1982. The long term effects of herbicide treatment on aquatic weed communities. In Proc. EWRS 6th Symp. on Aquatic Weeds. pp. 278-285.

Wade, P.M. and R.W. Edwards. 1980. The effect of channel maintenance on the aquatic macrophytes of the drainage channels of the Monmouthshire Levels, South Wales, 1840-1976. Aquat. Bot. 8:307-322.

Wagner, K. and H. Frehse. 1976. Methode zur Bestimmung von rüchständen des insektizids und akarizids^{R}. Folimat. Pflschutz-Nachir. Bayer 29:54-66.

Waite, D., H. Sommerstad, R. Grover and L.A. Kerr. 1984. South Saskatchewan watershed study (1984 interim report). Environmental Protection Service, Western and Northern Region. Environment Canada. 30 p.

Walker, B.H. and R.T. Coupland. 1968. An analysis of vegetation-environment relationships in Saskatchewan sloughs. Can. J. Bot. 46:509-522.

Walker, B.H. and R.T. Coupland. 1970. Herbaceous wetland vegetation in the aspen grove and grassland regions of Saskatchewan. Can. J. Bot. 48:1861-1878.

Walker, C.R.. 1962. Toxicological effects of several herbicides to bottom dwelling fish food organisms in Missouri ponds. Res. Rep. N. Cent. Weed Control Conf. 18th: 104-105.

Walker, C.R.. 1963. Endothal derivatives as aquatic herbicides in fishery habitats. Weeds 11:226-232.

Walker, C.R.. 1971. The toxicological effects of herbicides and weed control on fish and other organisms in the aquatic ecosystem. Proc. EWRS 3rd Internat. Symp. Aquatic Weeds. pp. 119-127.

Wallner, N.E., N.C. Leeling and M.J. Zabik. 1969. The fate of methoxychlor applied by helicopter for smaller European elm beetle control. J. Econ. Entomol. 62:1039-1042.

Walsh, G.E., C.W. Miller and P.T. Heitmuller. 1971. Uptake and effects of diclobenil in a small pond. Bull. Environ. Contam. Toxicol. 6(3):279-288.

Wan, M.T.K. and D.M. Wilson. 1976. The impact of mosquito control chemicals on selected non-target organisms in British Columbia. Environmental Protection Service Surveillance Report. EPS 5-PR-76-3. Environment Canada. 59 p.

Ware, G.W., E.P. Apple, W.P. Cahill, P.D. Gerhardt and K.R. Frost. 1969a. Pesticide drift. II. Mist-blower vs. aerial application of sprays. J. Econ. Entomol. 62(4):844-846.

Ware, G.W., N.A. Buck and B.J. Estesen. 1984. Deposit and drift losses from aerial ultra-low-volume and emulsion sprays in Arizona. J. Econ. Entomol. 77(2):298-303.

Ware, G.W., W.P. Cahill, P.G. Gerhardt and J.M. Witt. 1970. Pesticide drift. IV. On-target deposits from aerial application of insecticides. J. Econ. Entomol. 63(6):1982-1983.

Ware, G.W., B.J. Estesen, W.P. Cahill and K.P. Frost. 1971. Pesticide drift. V. Verticle drift from aerial spray applications. University of Arizona Agricultural Experimental Station Journal Series 1778.

Ware, G.W., B.J. Estesen, W.P. Cahill, P.D. Gerhardt and K.P. Frost. 1969b. Pesticide drift. I. High-clearance vs. aerial application of sprays. J. Econ. Entomol. 62(4):840-843.

Ware, G.W. and C.C. Roan. 1970. Interaction of pesticides with aquatic microorganisms and plankton. Residue Rev. 33:15-45.

Washino, R.K., W. Ahmed, J.D. Linn and K.G. Whitesell. 1972. Rice field mosquito control studies with low volume Dursban sprays in Coulsa County, California. Part 4. Effects upon aquatic non-target organisms. Mosq. News 32(4):531-537.

Water Pollution Research Laboratory. 1967. Control of aquatic weeds. pp. 40-44 In Water pollution research 1966. England.

Wauchope, R.D.. 1978. The pesticide content of surface water draining from agricultural fields, a review. J. Environ. Qual. 7(4):459-472.

Wauchope, R.D. and R.A. Leonard. 1980. Maximum pesticide concentrations in agricultural runoff: a semi-empirical prediction formula. J. Environ. Qual. 9(4):665-672.

Wauchope, R.D., L.E. Asmussen and E.W. Hauser. 1977. MSMA losses in runoff from small watersheds in the Mississippi Delta and Georgia coastal plain. Proc. South. Weed Sci. Soc. 30:395.

Way, J.M., J.F. Newman, N.W. Moore and F.W. Knaggs. 1971. Some ecological effects of the use of paraquat for the control of weeds in small lakes. J. Appl. Ecol. 8:509-532.

Weaver, G.D., M.J. Nilsson and R.E. Turney. 1982. Prospects for the prairie grain industry 1990. Canada Grains Council. Winnipeg, Canada. 305 p.

Weber, J.B.. 1972. Interaction of organic pesticides with particulate matter in aquatic and soil systems. In Fate of Organic Pesticides in the Aquatic Environment, S.D. Faust (Ed.). Adv. Chem. Ser. 111. pp. 55-120.

Weber, J.B. and H.D. Coble. 1968. Microbial decomposition of diquat adsorbed on montmorillonite and kaolinite clays. J. Agric. Food Chem. 16:475-478.

Webster, G.R.B. and G.W. Reimer. 1976. Field degradation of the herbicide metribuzin and its degradation products in a Manitoba sandy loam soil. Weed Res. 16:191-196.

Weed Society of America. 1983. Herbicide handbook of the Weed Society of America. Fifth Edition. Published by the Weed Society of America. Champaign, Illinois. 515 p.

Weidner, C.W.. 1974. Degradation in groundwater and mobility of herbicides. Report prepared for the Office of Water Research and Technology. U.S. Environmental Protection Agency, PB239242.

Weir, T.R. (Ed.). 1971. Atlas of the prairie provinces. Oxford University Press. Toronto.

Welch, J.L. and R.H. Ross. 1982. An approach to scoring of toxic chemicals for environmental effects. Environ. Toxicol. Chem. 1:95-102.

Welker, W.V. Jr. and D.N. Reimer. 1973. Control of fragant waterlily and spatterdock in irrigation ditches with glyphosate. Proc. Northeast. Weed Sci. Soc. 27:105-107.

Weller, M.W. 1981. Freshwater Marshes. Ecology and Wildlife Management. University of Minnesota Press, Minneapolis. 146 p.

Weller, M.W. and L.H. Fredrickson. 1973. Avian ecology of a managed glacial marsh. Living Bird 12:269-291.

Westerdahl, H.E. and J.F. Hall. 1983. Threshold 2,4-D concentrations for the control of eurasian watermilfoil and sago pondweed. J. Aquat. Plant Manage. 21:22-25.

Western Committee on Crop Pests. 1984. Report/Western Committee. B.D. Schaber and W.A. Charnetski (Eds.). Alberta, Canada.

Wetzel, R.G.. 1975. Limnology. W.B. Saunders Company. 743 p.

Whigham, D.F., J. McCormick, R.E. Good and R.L. Simpson. 1978. Biomass and primary production of freshwater tidal marshes. In Freshwater Wetlands: Ecological Processes and Management Potential, R.E. Good, D.F. Whigham and R.L Simpson (Eds.). Academic Press. pp. 3-20.

White, A.W., L.E. Asmussen, E.W. Hauser and J.W. Turnbull. 1976. Loss of 2,4-D in runoff from plots receiving simulated rainfall and from a small agricultural watershed. J. Environ. Qual. 5:487-490.

White, A.W., A.D. Barnett, B.G. Wright and J.H. Holladay. 1967. Atrazine losses from fallow land caused by runoff and erosion. Environ. Sci. Technol. 1:740-744.

White, A.W. Jr., L.A. Harper, R.A. Leonard and J.W. Turnbull. 1977. Trifluralin volatilization losses from a soybean field. J. Environ. Qual. 6:105-110.

Whittaker, R.H.. 1975. Communities and ecosystems. MacMillan Publishing Co.. 385 p.

Wilce, S.E., N.B. Akesson, W.E. Yates, P. Christensen, R.E. Cowden, D.C. Hudson and G.I. Weigt. 1974. Drop size control and aircraft spray equipment. Agric. Aviation 16(1):7-13.

Wilkinson, W. and F.D. Smith. 1977. PP557: ecological effects in three ponds, U.K., 1976-3. Fish and invertebrates. ICI Plant Protection Division Rep. No. TMJ1475B, July 1977.

Williams, P.P.. 1977. Metabolism of synthetic organic pesticides by anaerobic microorganisms. Residue Rev. 66:63-135.

Williams, E.H., E.L. Mather and S.M. Carter. 1984. Toxicity of the herbicides endothall and diquat to benthic crustaceans. Bull. Environ. Contam. Toxicol. 33:418-422.

Willis, A.J.. 1972. Long-term ecological changes in sward composition following application of maleic hydrazide and 2,4-D. In Proc. 11th British Weed Control Conf. British Crop Protection Council. pp. 360-367.

Willis, G.H., R.L. Rogers and L.M. Southwick. 1975. Losses of diuron, linuron, fenac and trifluralin in surface drainage water. J. Environ. Qual. 4:399-402.

Wilson, D.C. and C.E. Bond. 1969. The effects of the herbicides diquat and dichlobenil (Casoron^R) on pond invertebrates. Part I. Acute toxicity. Trans. Amer. Fish. Soc. 3:438-443.

Wojtalik, T.A., T.F. Hall and L.O. Hill. 1971. Monitoring ecological conditions associated with wide-scale applications of DMA 2,4-D to aquatic environments. Pestic. Monit. J. 4(4):184-203.

Wolfe, N.L., R.G. Zepp, J.A. Gordon, G.L. Baughman and D.M. Cline. 1977a. Kinetics of chemical degradation of malathion in water. Environ. Sci. Technol. 11(1):88-93.

Wolfe, N.L., R.G. Zepp, D.F. Paris, G.L. Baughman and R.C. Hollis. 1977b. Methoxychlor and DDT degradation in water: rates and product. Environ. Sci. Technol. 11(12):1077-1081.

Wolfe, N.L., R.G. Zepp and D.F. Paris. 1978. Carbaryl, propham and chlorpropham: a comparison of the rates of hydrolysis and photolysis with the rate of biolysis. Water Res. 12:565-571.

Womeldorf, D.J., P.A. Gilles and K.E. White. 1972. Insecticide susceptibility of mosquitoes in California illustrated distribution of organophosphorus resistence in larval Aedes nigromaculis and Culex tarsalis. Proc. Pap. Ann. Conf. Mosq. Control Assoc. 40:17-21.

Woodward, D.F. and W.L. Mauck. 1980. Toxicity of five forest insecticides to cutthroat trout and two species of aquatic invertebrates. Bull. Environ. Contam. Toxicol. 25:846-853.

Worthing, C.R. and S.B. Walker. 1983. Pesticide manual. British Crop Protection Council. Lavenham Press Limited,. Lavenham, Suffolk, England. 695 p.

Wright, S.J.L.. 1972. The effect of some herbicides on the growth of Chlorella pyrenoidosa. Chemosphere 1:11-14.

Yates, W.E., N.B. Akesson and D.E. Bayer. 1978. Drift of glyphosate sprays applied with aerial and ground equipment. Weed Sci. 26(6):597-604.

Yates, W.E., N.B. Akesson and R.W. Brazelton. 1981. Systems for the safe use of pesticides. Outlook Agric. 10:321-326.

Yates, W.E., N.B. Akesson and H.H. Coutts. 1966. Evaluation of drift residues from aerial applications. Trans. ASAE 9:389-397.

Yemm, E.W. and A.J. Willis. 1962. The effcets of maleic hydrazide and 2,4-dichlorophenoxyacetic acid on roadside vegetation. Weed Res. 2:24-40.

Yeo, R.R.. 1967. Dissipation of diquat and paraquat and effects on aquatic weeds and fish. Weeds 15:42-46.

Yoo, J.Y., D.C.G. Muir and B.E. Baker. 1981. Persistence and movement of cyanazine and procyazine under field conditions. Can. J. Soil Sci. 61:237-242.

Yoshioka, H.. 1978. Development of fenvalerate a new and unique synthetic pyrethroid containing the phenyl isovaleric-acid moiety. Rev. Plant Prot. Res. 11:39-52.

Yu, C.C., G.M. Booth and J.R. Larsen. 1975a. Fate of triazine herbicide cyanazine in a model ecosystem. J. Agric. Food Chem. 23:1014-1015.

Yu, C.C., D.J. Hansen and G.M. Booth. 1975b. Fate of dicamba in a model ecosystem. Bull. Environ. Contam. Toxicol. 13:280-283.

Zepp, R.G. and D.M. Cline. 1977. Rates of direct photolysis in aquatic environments. Environ. Sci. Technol. 11:359-366.

Zepp, R.G., N.L. Wolfe, J.A. Gordon and G.L. Baughman. 1975. Dynamics of 2,4-D esters in surface waters: hydrolysis, photolysis and vaporization. Environ. Sci. Technol. 9:1144-1150.

Zinkl, J.G., C.J. Henney and P.J. Shea. 1979. Brain cholinesterase activities of passerine birds in forests sprayed with cholinesterase inhibiting insecticides. In Animals as Monitors of Environmental Pollutants. National Academy of Sciences. pp. 356-365.

Zinkl, J.G., J. Rathert and R.R. Hudson. 1978. Diazinon poisoning in wild Canada geese. J. Wildl. Manage. 42:406-408.

Zinkl, J.G., R.B. Roberts, C.J. Henney and D.S. Lenhart. 1980. Inhibition of brain cholinesterase of forest birds and squirrels exposed to aerially applied acephate (Orthene^R). Bull. Environ. Contam. Toxicol. 24:676-683.

Zinkl, J.G., R.B. Roberts, P.J. Shea and J. Lasmanis. 1981. Toxicity of acephate and methamidophos to dark-eyed juncos. Arch. Environ. Contam. Toxicol. 10:185-192.

Zitko, V.. 1980. Relationships governing the behavior of pollutants in aquatic ecosystems and their use in risk assessment. Can. Tech. Rep. Aquat. Sci. no. 973. pp. 243-265.

Zitko, V. and D.W. McLeese. 1980. Evaluation of hazards used in forest spraying to the aquatic environment. Can. Tech. Rep. Fish. Aquat. Sci. no. 985. pp. iii-21.

Zitko, V., D.W. McLeese, C.D. Metcalfe and W.G. Carson. 1979. Toxicity of permethrin, decamethrin, and related pyrethroids to salmon and lobster. Bull. Environ. Contam. Toxicol. 21:338-343.

Zittlau, W.T.. 1979. An environmental assessment of agricultural practices and policies: implications for waterfowl habitat management. Natural Resources Institute. University of Manitoba, Winnipeg, Manitoba. 282 p.